NORTHERN TERRITORY NATURALIST

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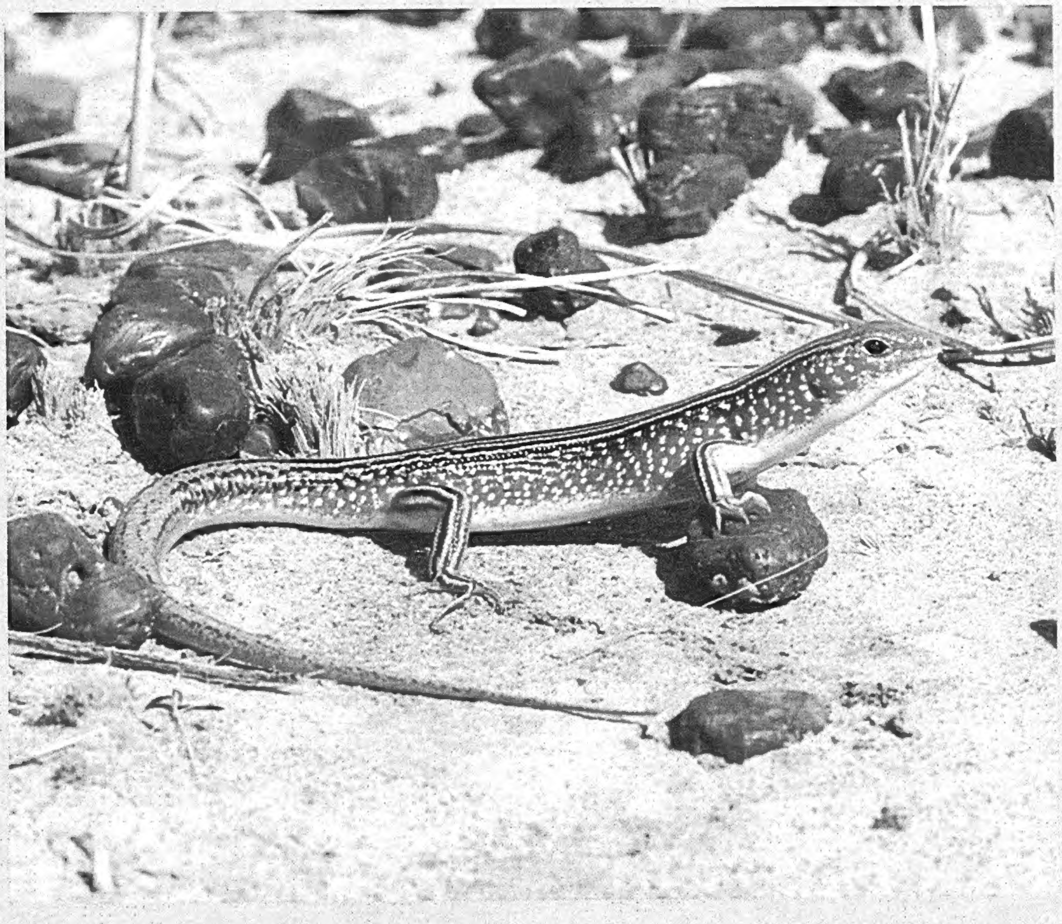

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Front Cover: Pretty Ctenotus *(Ctenotus pulchellus)* removed from pipeline trench and released, Barkly Tablelands, Northern Territory. (Christy Harvey)

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Number 29 **July 2019**

Contents

Understanding Australian tropical savanna: environmental history from a pollen perspective

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Abstract

Understanding the long-term functioning of Australia's tropical savannas is central to the management and conservation of these ecosystems. An environmental history of the Darwin region's mesic savanna is presented from Girraween Lagoon, approx. 25 km southeast of Darwin, where pollen and charcoal analysis of a 5 metre sediment core provides a record spanning the previous 12,700 years. Results show the gradual development of permanent water at the site, surrounded by a dynamic landscape where changing climates and local people's use of fire has shaped the vegetation from that of a savanna to an open forest.

Introduction

Palaeoecology is the branch of ecology that studies past ecosystems and their trends in time using fossils and other proxies (Rull 2010). These studies hold major implications with respect to our understanding of the nature and function of contemporary ecosystems, contributing to key questions of natural variability, baselines, thresholds, resilience, and regime shifts (Froyd & Willis 2008). In this respect, palaeoecology and other forms of long term science are central to land resource management, and play an increasingly important role informing climate change conservation strategies (Ekblom & Gillson 2017).

Palynology is the scientific discipline concerned with the analysis of plant pollen, spores, and certain microscopic planktonic organisms, in both living and fossil form (Jarzen *et al.* 1996). As pollen and spores are produced in large numbers, then dispersed over wide areas by wind and water, and because they are strongly resistant to decay under certain conditions (i.e. anaerobic environments), their fossils are recoverable from a variety of sedimentary materials (Brown 2008). Birks (2012: 292) described records of pollen preserved in lake or swamp sediments as "long-term ecological observatories". As an indicator of past vegetation communities, including relations between plants and their environments, palynology is one of the most important tools for palaeoecology.

Darwin's Girraween Lagoon (12.517°S, 131.081°E) (Figure 1a) is a site of ongoing palaeoecological investigation. It forms one of a series of research locations from the Top End of the Northern Territory. These sites are focused on the development of a concentrated spatial and temporal network of Quaternary records of vegetation and environmental change in the savanna of tropical northern Australia. The Quaternary is the most recent geological period of time, spanning the past two million years up to the present day. It is subdivided into the Pleistocene (glacial phase) and Holocene (present interglacial) epochs, with the Holocene covering the past 11,700 years (Cohen *et al.* 2013; updated). The savanna region of northern Australia remains little studied in the Quaternary context; i.e. fine-resolution palaeoecology and palynology, including detailed chronologies of vegetation and fire regime change are scarce, particularly for the Northern Territory. We seek to document changes before, during and after the arrival of humans (Indigenous and European). We further seek to track changes across major glacial and interglacial climatic cycles. We are asking questions such as 'What are the major drivers of ecosystem change at different time scales?' and 'What determines landscape and habitat diversity over time?' We aim to explore whether the Australian

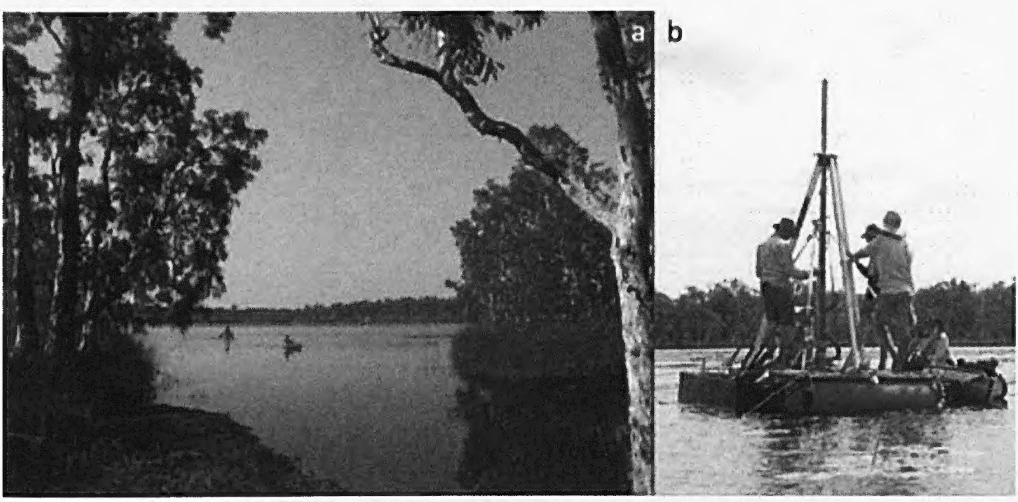

Figure 1. (**a**) View of Girraween Lagoon taken from the northern shoreline with fringing *Melaleuca* trees in the foreground. (**b**) Floating platform with hydraulic coring rig used to sample lagoon sediment. August 2018. (Cassandra Rowe)

savanna represents a region in long term transition or a region in a stable ecosystem state. In this project, Girraween is representative of a mesic tropical savanna (>1200 mm/yr of rainfall (Russell-Smith *et al.* 2010)), and following the regional 'savanna' land cover classifications of Hutley *et al.* (2013) and Moore *et al.* (2018). Moore *et al.* (2016) refer to the wider Howard Springs area as an open-forest savanna.

Methods

In September 2015, Girraween Lagoon was cored using a floating platform with a hydraulic coring rig (Figure 1b). A 19.4 m long core was collected in 1 m sections to bedrock. The focus of this report is a summary of the upper 5 m of this core, encompassing the last 12,700 years. The core sections were split in half, described, and subsampled at 10 cm intervals. Two cubic centimetre sediment samples were processed for pollen (see Bennett & Willis (2001)). Charcoal incorporated within this final sediment concentrate was counted simultaneously with pollen as an indicator of landscape fire, as guided by Whitlock & Larsen (2001). Seven bulk sediment samples were pre-treated for radiocarbon dating according to Bird *et al.* (2014) and sent to Australia's Nuclear Science and Technology Organisation for processing. Radiocarbon dating is one of the most widely used scientific dating methods in environmental science and archaeology today. Radiocarbon measurements are reported in terms of years 'before present' (BP), but are not a true calendar age and therefore require calibration. Once it is calibrated to a calendar time scale, a radiocarbon date is expressed as 'cal BP' (Ramsey 1995). In this study, age reporting follows Stuiver & Polach (1977), converted into calibrated ages (cal BP) using CALIB REV7.1.0 (Stuiver & Reimer 1993, Hogg *et al.* 2013; calibration curve SHCal13). All data were graphed using TGView (Grimm 2004).

Because discrimination of pollen to the level of species is often difficult, the taxa are only given at genus level in the following text (e.g. *Casuarina* rather than *Casuarina equisetifolia*).

Results

Results span the Holocene timeframe. A summary pollen diagram with charcoal results is presented in Figure 2. Pollen diagrams are a form of graph. The Y-axis shows age (time) with core depth. The X-axis shows the percentage of plant pollen types; with greater graph width for example, the larger the percentage of a given pollen (plant) type present. Changes in the percentages of plant types are displayed as the core progresses with time (depth) and are interpreted to show similar changes in vegetation composition (compare with Faegri *et al.* 2000).

In total, 117 pollen types were identified. They were divided into 11 summary groups to capture plant form and/or vegetation type:

- Myrtaceae associated with dryland environments (10 taxa, e.g. eucalypts and *Calytrix*);
- Myrtaceae associated with wetland environments (8 taxa, e.g. *Melaleuca* and *Leptospermum*);

- other sclerophyll pollen (20 taxa);
- monsoonal forest associates (22 taxa);
- lianes and mistletoes (6 taxa);
- mangrove (or coastal, 3 taxa);
- grasses (Poaceae) (2 grain size classes);
- herbaceous plants (29 taxa, including sub-shrubs);
- sedges (and similar wet-ground taxa, 7 taxa);
- aquatics (3 taxa);
- pteridophytes (7 taxa).

Figure 2 shows some representative pollen grains identified from the core at Girraween Lagoon

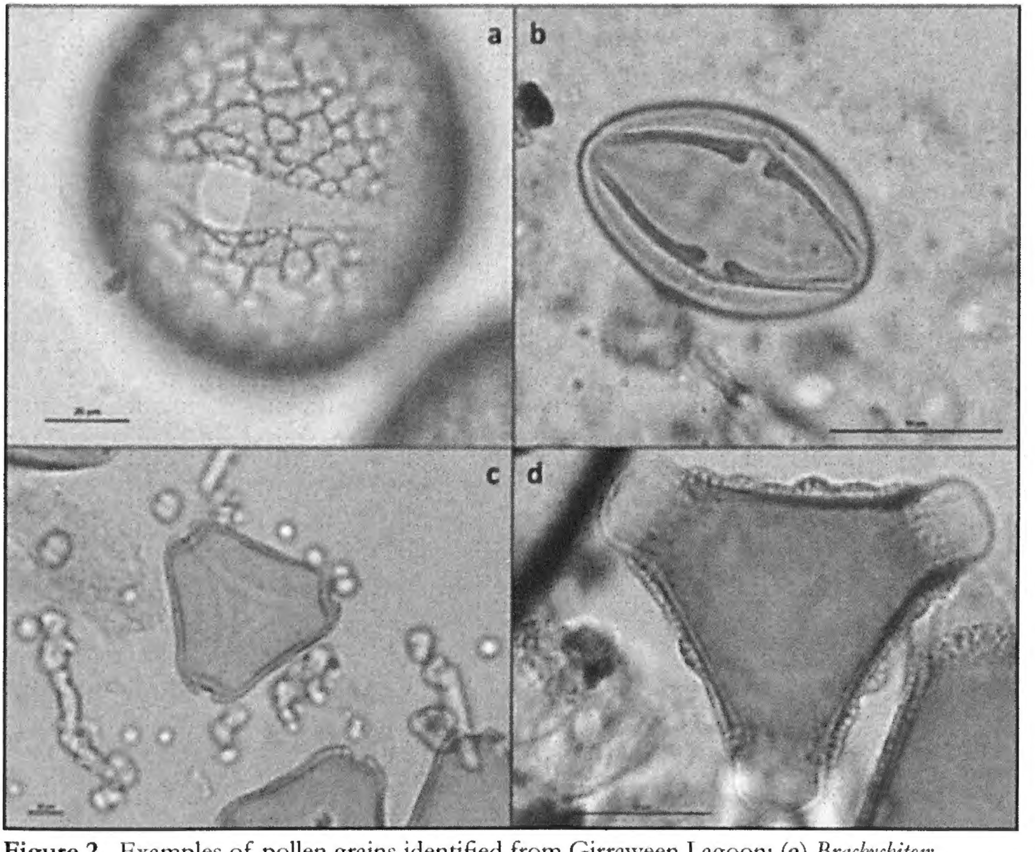

Figure 2. Examples of pollen grains identified from Girraween Lagoon: (**a**) *Brachychiton*; (**b**) *Buchanania*; (**c**) *Eucalyptus*; (**d**) *Grevillea*). (Cassandra Rowe)

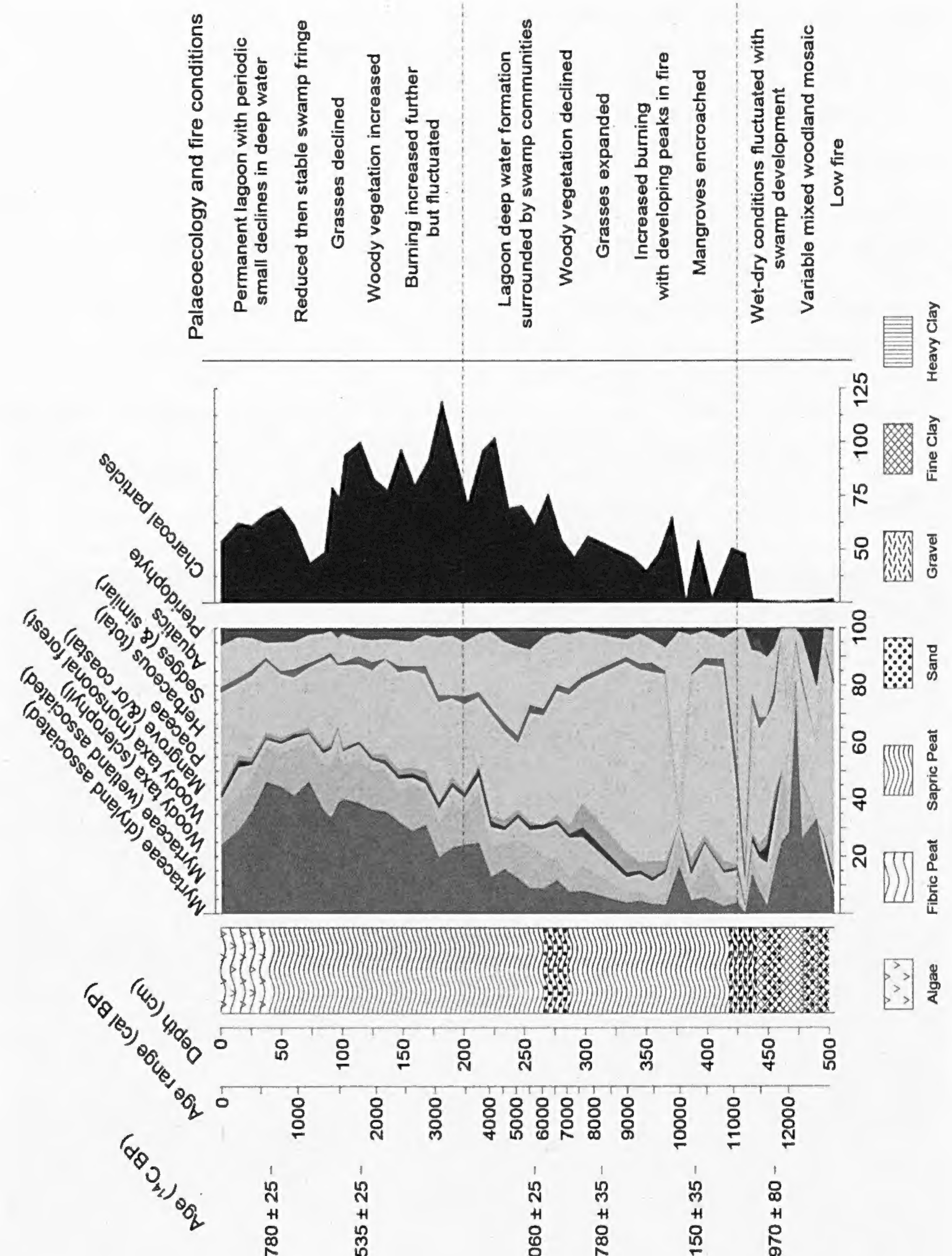

Figure 3. Pollen diagram and palaeoecological interpretation showing summary plant groups (percentage) and fire history (i.e. charcoal accumulation rate) plotted against depth, sediment type, and radiocarbon and calibrated age.

Discussion

Mangrove and coastal pollen

This study recorded *Rhizophora*, *Ceriops* and *Casuarina* pollen types. These pollens are wind-distributed and, as such, can travel long distances (Rowe 2012). The plants themselves were not present on-site at Girraween during the Holocene, but responded to the main phases of sea level rise and fall as documented for the northern Australian coast (Chappell 2001). Mulrennan & Woodroffe (1998), for example, demonstrate the development of widespread mangrove swamps following saltwater intrusion into creeks and rivers up to mid-Holocene times in the Northern Territory.

Marine mangroves approached the Girraween site between 11,700–10,800 cal BP, and reached their closest point between 9700–7100 cal BP. After 7100 cal BP, mangroves contracted coastward and remained stable in the broader Darwin area until 2150 cal BP. They declined further beginning 1950 cal BP, but recovered within the previous 1000 years to present day regional coverage. *Casuarina* also encroached on Girraween in the early Holocene but reduced and fluctuated as the mangrove forests expanded.

Lagoon development and climatic observations

Today's open water lagoon is approximately 6100 years old (pers. obs). Prior to this time, Girraween was a patchy swamp and its environment reflected drier climate transitions at the end of the last glacial period, including an intermittent (inactive) monsoon (Reeves *et al.* 2013). Between 12,700 and 10,200 cal BP the site fluctuated between wet and dry conditions. Sedges and ferns expanded and contracted in alternate phases with grasses and herbs. Aquatic plants (such as the waterlily taxa *Nymphoides* and *Nymphaea*) were absent, highlighting irregular and/or weak wet seasonality and no permanent standing water. Tree growth, such as that of *Melaleuca*, around Girraween's swampy areas was sparse at this time.

After 10,200 cal BP, the water at Girraween began to pond. Shallow waters expanded from 9500 cal BP, followed by slower increases in deeper water. Both *Nymphoides* and *Nymphaea* were present, but only became fully established around 6100 years ago, indicating permanent open water with a range of depths and with greater lagoon stability. Helping the lagoon to develop, the climate at this time exhibited highest regional rainfall and higher temperatures, with reduced annual seasonality (Reeves *et al.* 2013). It is from this point onwards that water conditions comparable to the modern site existed. Permanent, deeper water zones show some subsequent decline, at times 3750–3500 cal BP, 2850 cal BP and 1300–1250 cal BP.

As Girraween gradually formed permanent open water, swamp habitats rearranged into an encircling fringe. Sedge communities expanded first, to become most extensive between 6050–4050 cal BP. Wetland woodlands became more prominent from 5500–5000 cal BP. *Melaleuca* dominated, with *Leptospermum* and *Asteromyrtus* in the subcanopy. *Pandanus* was also present. These woodlands remained stable, however the sedge swamp

undergrowth declined (narrowed) beginning 2850 cal BP, then expanded again in the last 350 years reaching modern day distributions.

Water level changes during the last 4000 years are consistent with increasing climatic variability across northern Australia toward the present day (e.g. weakening of monsoon rainfall (Denniston *et al.* (2013), enhanced seasonality, and emerging El Niño-Southern Oscillation relationships (McGowan *et al.* (2012)). Past extent and impact of changing monsoon activity on wetlands, including drying/arid phases in the late Holocene, can be seen by comparing Girraween with similar palaeoecological studies from northern Western Australia's Kimberley region (e.g. Field *et al.* (2017)).

Savanna transformation and fire in the landscape

By 12,700 years ago a changeable mixed wooded-savanna was present at Girraween. Just as the wetland was fluctuating between wet and dry in the early parts of the record, the surrounding catchment vegetation was also variable and patchy. Eucalypts were dominant with other sclerophyll and forest-associated taxa incorporated into the subcanopy. The pollen shows a number of (semi-) deciduous tree taxa (e.g. *Terminalia, Bombax* and *Canarium*) suited to inconsistent moisture supply from variability in rainfall pattern. It is thought that alternating peaks in different plant communities did not initially support extensive burning; that patchy vegetation patterns and changeable biomass hindered fire spread across the catchment. The climate at this time may also have been less amenable to burning.

Beginning 11,800 cal BP, the trees and shrubs declined at the site, particularly the eucalypts. This resulted in an extended phase of open vegetation at Girraween 11,700–5500 cal BP. As the eucalypts decreased, grasses expanded and an open savanna developed. Grass cover was pronounced between approximately 10,800 cal BP and 5500 cal BP, and in the period 10,200–8900 cal BP maintained maximum distribution across the catchment. Despite a wetter-warmer climate, grass competition and small grass-fed fires between 10,800 cal BP and 5500 cal BP helped keep the woody plant presence low. Gradual tree-shrub expansion and the beginning of grass decline occurred, as charcoal records show increased burning and the start-appearance of small peaks in fire. This study links fire extent to the abundance of woody biomass at Girraween.

Grass cover decreased compared to woody cover after 4500 cal BP. At the same time, burning further expanded and there were major peaks in fire occurrence. A higher level and ongoing burning-type fire regime appeared to support woody plant proportions in the canopy and subcanopy. Increased climatic variability, including drying conditions, would have encouraged fire. However, this charcoal change is also consistent with the age of archaeological evidence demonstrating local expansions in indigenous occupation and use of the area (Bourke 2004; Brockwell 2005). Large water sources are also described as locations where hunter-gatherer populations concentrated (Williams *et al.* 2015). Intentional landscape burning by indigenous people is therefore one of the interacting variables which shaped higher and more consistent fire regimes in the

late Holocene. Eucalypts initially expanded between 4100 and 3100 cal BP. Maximum sustained eucalypt pollen spans 2850–600 cal BP, indicating an open forest savanna, and expanded further through the period 1150–600 cal BP. Shrub-herbaceous taxa also increased in this period, adding to ground layer diversity amidst less grasses. Pollen and charcoal results suggest this was a stable open forest savanna system, in line with an environment managed by local people to maintain resources in the face of variable climate and a declining monsoon. Today, remnant open forest conditions remain in place, partly maintained by Landcare groups as urban bushland and an area of recreational use (e.g. boating). Looking into the future, reports such as those by Brock (1995), Lamche (2008) and Lamche & Schult (2012) foresee management issues for the lagoon and the surrounding vegetation, noting impacts through fragmentation and weeds in particular.

Conclusion

Change is a continuous process in the Northern Territory's Top End environments, including the Darwin region. This paper introduces Girraween Lagoon as a site of Quaternary study, and the palaeoecological/palynological techniques helping to highlight the nature of long-term savanna dynamics, and the varied outcomes from vegetation-fire-climate interactions. The Girraween catchment experienced early alternating periods of waterlogging and temporary swamp development, prior to permanent lagoon formation approximately 6100 years ago. Simultaneously, the surrounding catchment transformed from a grassy savanna to denser woody savanna and forest, with changes in the use of fire playing a particular role in the vegetation becoming increasingly dense (wooded). These results provide an example of the baseline information available to contemporary ecologists and management/conservation personnel, and are the start of further research across the region.

Acknowledgements

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Effects of soil treatments on the establishment of *Acacia* and *Eucalyptus* following gravel extraction

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Abstract

Sources of construction materials such as gravel are sought near urban areas, including near Darwin, and these sites are required to be rehabilitated. However, few studies have investigated techniques to improve seedling establishment on gravel rehabilitation sites in northern Australia. This study investigated seedling establishment of the locally dominant tree species, Darwin Stringybark (*Eucalyptus tetrodonta*), Darwin Woollybutt (*E. miniata*) and the shrub, Western Salwood (*Acacia lamprocarpa*), on a gravel rehabilitation area near Howard Springs and in a shade house at Charles Darwin University, Darwin. The aim was to determine the effects of six treatments, including scarification, topsoil and application of leaf litter, and combinations of these treatments, on the emergence and survival of *Acacia* and *Eucalyptus* seedlings. Seedling establishment on the compacted gravel substrate was minimal, whereas application of litter or scarification of the gravel surface was found to increase seedling emergence and establishment. Seedling emergence was much greater in the shade house for all species (41%) than under field conditions (8%) suggesting that water stress was likely the major factor affecting survival in the field. Greater wind disturbance, seedling herbivory and erosion could have been additional factors. However, the application of topsoil had relatively little benefit for emergence, although later survival of *E. miniata* seedlings in topsoil was high. We suggest that altering or modifying the microsites of the gravel extraction areas by relatively easy and cost-effective methods such as scarification and application of litter can improve the seedling establishment of native woody species and help in rehabilitation of gravel extraction sites. Application of litter and scarification of the substrate are likely to enhance water availability for seedlings, possibly by reducing heating and evaporation, and by increasing water infiltration.

Introduction

Demand for construction materials such as gravel is increasing globally. Worldwide, the annual demand for construction aggregates in 2019 is estimated at over 51 billion tonnes (Freedonia 2016). Local demand for construction materials is substantial and has resulted in clearing of approximately 17 ha of native vegetation per year in the Darwin region for gravel extraction (Price *et al.* 2005). In the Darwin region, gravel is extracted from shallow borrow pits, after initial clearing of vegetation and stockpiling of the

upper soil layer (Setterfield *et al.* 1993). In comparison to unmined sites, these processes can reduce the moisture-holding capacity, reduce nutrient (especially available nitrogen) availability and increase penetration resistance to root growth due to surface compaction (Johnson 1987; Schwenke *et al.* 1999). Even when fertiliser is added, nutrients are easily leached from the sites (Johnson 1987) and lateritic soils can have great capacity for immobilisation of phosphorus making it unavailable for plant growth (Bell 1985). Alterations to the hydrology following gravel extraction also have substantial impacts on the vegetation (Johnson 1987; Setterfield *et al.* 1993). A study in the Czech Republic by Rehounkova & Prach (2006) found that hydrology was the most important local site factor influencing the course of vegetation succession, but other significant influences were surface soil texture, soil pH and the presence of nearby communities containing desired species.

Unaided natural revegetation of gravel borrow pits is typically slow and generally unsuccessful. Setterfield *et al.* (1993) found that gravel borrow pits for road construction in Kakadu National Park recovered very little compared to their original vegetation. Similarly, Price *et al.* (2005) found that recovery of vegetation in the Darwin region was poorer in gravel mines than in sand mines. They found that only 41% of native vegetation species recovered in gravel mine sites compared to the species richness in control sites, with only 2% recovery of tree density, although the sites were up to 27 years old. It is very rare that the locally dominant *Eucalyptus* species re-establish in gravel extraction sites (Setterfield *et al.* 1993), although more recent observations suggest some *Eucalyptus* have established (Chris Brady pers. comm.). The extent to which the disturbed sites eventually recover their ecological character after gravel extraction is unknown. Poor natural establishment can be due to physical and chemical changes to the substrates, lack of nutrients, and limited or excessive water (Johnson 1987; Corbett 1999). Even in undisturbed soils of the wet-dry tropics, low levels of nitrogen and phosphorus limit the growth of native species (Bell 1985) and this is exacerbated by loss, disturbance or stockpiling of topsoil (Schwenke *et al.* 1999). In such circumstances, research on rehabilitation is essential for better outcomes, yet most research has focused on rehabilitation of larger mining sites; very little research has focused on the rehabilitation of gravel mine sites.

For most plant species, recruitment is limited primarily by microsite availability rather than seed availability (Crawley 1990). Microsite conditions at a small scale affect seed germination and seedling establishment and this is particularly important for small-seeded species such as *Eucalyptus* (Battaglia & Reid 1993). Important characteristics of microsites which can affect seed germination and seedling establishment include topographical features, soil texture, amount of litter, and presence of rocks or woody debris (Eldridge *et al.* 1991; Oswald & Neuenschwander 1993; Titus & del Moral 1998). *Eucalyptus* seedling establishment has been found to be affected by various factors including soil moisture, drought, temperature, light, overstorey shading, canopy gap, litter accumulation, grazing, plant competition, and insect and fungal attack (Setterfield

et al. 1993; Li *et al.* 2003). Litter cover effects on *Acacia*, *Corymbia* and *Eucalyptus* can vary with species and soil type (Saragih & Bellairs 2015). In addition to that, disturbances like fire, grazing, cultivation, addition of fertiliser and soil compaction (or their interactive/combined effects) also affect the establishment of eucalypt seedlings (Li *et al.* 2003). Small scale variation in soil conditions, even at the scale of tens of centimetres, can affect germination and establishment of *Eucalyptus* (Battaglia & Reid 1993). At older mine sites that had similar grass cover to that of the unmined savanna sites and where litter cover was substantial, eucalypt establishment can be successful (Saragih *et al.* 2015). Demonstrating this, across 14 gold mine rehabilitation sites south of Darwin, the average density of *Eucalyptus*/*Corymbia* seedlings in 12 year old mine rehabilitation sites was 528 ± 131 seedlings ha^{-1} (Saragih *et al.* 2015).

The aim of this study was to investigate whether seedling establishment of common Northern Territory savanna tree and shrub species can be improved by treatments to alter and create establishment microsites in gravel extraction areas. Specifically, the effects of scarification of the gravel surface, application of litter and application of soil and their combined effects on seedling emergence and growth of Darwin Stringybark (*Eucalyptus tetrodonta*), Darwin Woollybutt (*E. miniata*) and Western Salwood (*Acacia lamprocarpa*) were investigated in a gravel extraction site in the field and also in a shade house at Charles Darwin University.

Methods

The field trial was carried out at Scrubby Creek in the Howard Springs region (12.794 °S, 131.224 °E), 31 km east of Darwin. The site was formerly *Eucalyptus tetrodonta* dominated savanna woodland that had been cleared for gravel extraction. Trees and shrubs were cleared and burnt, then the top 0.2 m of topsoil and remaining vegetation were stockpiled (Keith Joy pers. comm.). After extraction of the gravel, the topsoil was spread out in 0.3 m high heaps over the site, enabling unaided vegetation recruitment to occur.

Three common local woody savanna species were chosen for the study: *Eucalyptus miniata*, *E. tetrodonta* and *Acacia lamprocarpa*. The two *Eucalyptus* are dominant overstorey trees while *Acacia lamprocarpa* is a large shrub to small tree growing from 4–12 m high. Seeds of all three species were obtained from Greening Australia.

Seed germination testing using four replicates of 25 seeds per species was carried out. Each replicate was placed in a plastic 9 cm diameter Petri dish lined with two filter papers soaked with 70 ml of 0.08 gm L^{-1} of Banrot 400 WP Fungicide. For *Acacia lamprocarpa*, four replicates of 25 seeds were placed in boiling water for 1 minute before being placed in Petri dishes like the the other seeds. The Petri dishes were incubated at 30 °C in a controlled temperature incubator with a 12 hour light/dark regime and monitored for germination three times a week for four weeks. Germination (presence of an emergent radical) was recorded. Germinated and dead (soft seeds showing discoloration or decomposition of tissue) seeds were removed and percent germination was calculated after four weeks.

Seeds of the three species were sown into 0.5 x 0.4 m plots, with various substrate conditions, at the Scrubby Creek field site on 7 March 2018. Rainfall included 56 mm on 5 and 6 March on the days prior to planting, 143 mm from 7–31 March, 78 mm in May, and 0 mm in June (Howard Springs rainfall data from www.bom.gov.au). The control was the bare gravel base, 'Litter' had 122 gm of local *Eucalyptus* woodland litter applied on the gravel surface, 'Topsoil' had 0.02 m depth of topsoil placed on the gravel, and 'Scarified' had the surface broken up to 0.02 m depth with a mattock. Other treatments were combinations, with 'Scarified plus litter' and 'Topsoil plus litter'. Fifty seeds were sown for each treatment within a 0.4 m x 0.3 m area in the middle of the plot (Figures 1, 2). *Eucalyptus miniata* and *Acacia lamprocarpa* had six replicates per treatment. *Eucalyptus tetrodonta* had only sufficient seeds for the control, litter, scarified and topsoil treatments and for 25 seeds per plot.

Similar treatments were applied in a shade house at Charles Darwin University. Gravel from the Boral Quarry to the north of Gunn Point Road was used. Plastic trays (0.42 x 0.32 x 0.20 m depth) had six holes drilled to allow drainage. Gravel was hardened by compressing and moistening simultaneously to make a compact hardpan substrate similar to that of the field. After making the hardpan, the same treatments were applied as for the field site; that is, litter was applied at an equivalent rate of 70 gm per tray, the top 0.02 m was scarified using a hand cultivator, and 0.02 m of topsoil from the field site was overlaid. The same combinations of treatments were also applied as with the

Figure 1. Overview of one of the treatment areas on the gravel rehabilitation field site, with small mounds of topsoil surrounding the treatment area applied as part of general rehabilitation of the site. (Sean Bellairs)

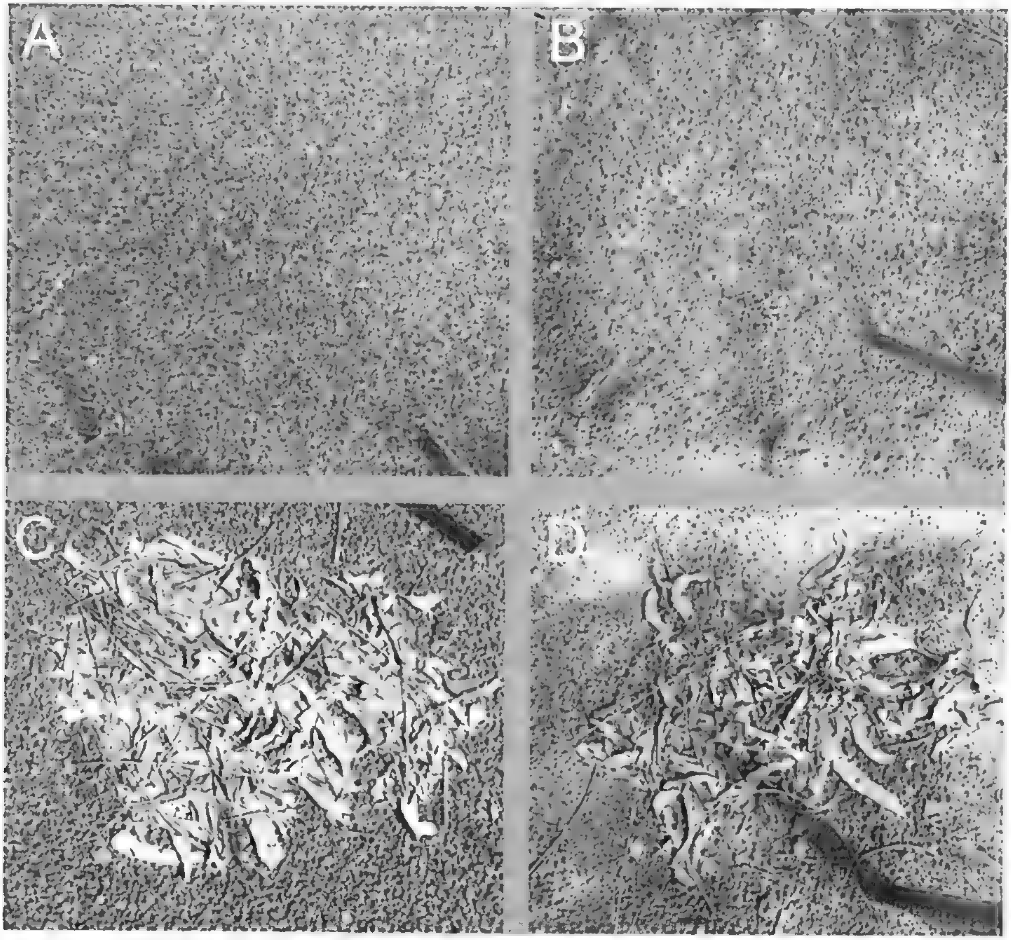

Figure 2. Photos of the treatments in the field site one month after application, including: (**A**) the bare gravel base, (**B**) topsoil, (**C**) leaf litter and (**D**) scarification plus leaf litter. (Sean Bellairs)

field trials. *Eucalyptus miniata* and *Acacia lamprocarpa* had six replicate trays per treatment with 50 seeds per tray. *Eucalyptus tetrodonta* had only the control and gravel plus topsoil treatments, with only four replicates and 25 seeds per tray due to limited seed availability.

Seeds were placed on the surface of gravel, scarified gravel and soil, whereas seeds were placed before the litter was applied in the litter treatments. The field experiment relied on natural rainfall. Irrigation in the shade house was applied three times a day for 10 minutes. The number of seedlings that emerged was observed each fortnight for two months in the field until May 2018. The seedlings were recorded in the field, with a photograph taken of each plot on each monitoring occasion. In the shade house, seedling emergence was recorded three times a week for two months. Seedlings were marked with toothpicks and dead plants were recorded. Up to eight seedlings per tray or plot were randomly selected and had their height (from shoot apical meristem to the surface of the substrate) measured at the end of the experiment.

Data analysis for seedling emergence in the field trial involved each species being separately analysed using GLM with a binomial distribution. The data were binomial, as each of the 50 seeds could either emerge or not by the end of the trial. The treatments were fixed effects, and block was included as a factor. The gravel control was the baseline for the intercept in the models. Multiple comparisons of means were analysed using Tukey contrasts to identify differences between individual pairs of treatments.

Seedling emergence in the shade house was analysed similarly, except there were no blocks and, to account for over-dispersion, a quasibinomial distribution was used. Analyses were performed using the packages 'lme4' and 'MASS' version 7.3-47 in the R environment 3.4.1 (R Core Team 2017), with a significance level of $P \leq 0.05$.

For plant height analysis of field treatments, GLM was used to allow the unbalanced design with different numbers of plants per treatment. For plant height analysis of shade house treatments, ANOVA was used, as the number of plants used for height measurement in each treatment was relatively uniform.

Results

Untreated seeds of *Acacia lamprocarpa* had 32 ± 3% germination, and boiling the seeds reduced germination; therefore the seeds used in the subsequent trials were not boiled. Germination of *Eucalyptus tetrodonta* was 44 ± 20% and *E. miniata* was 38 ± 11%. The *Acacia* had 96% of seeds germinate within 17 days and the *Eucalyptus* species had all seeds germinate within 9 days.

In the shade house, application of topsoil and litter on the gravel resulted in 57 ± 6% emergence, which was significantly higher than the scarified gravel treatment or compacted bare gravel (Figure 3). Application of litter, or litter and topsoil, increased emergence of *A. lamprocarpa* to above 40%, which was significantly greater than the 15 ± 4% emergence of the seeds sown onto compacted bare gravel. Mean emergence from the scarified gravel treatment at 28 ± 5% was not significantly different to the gravel or scarified gravel treatments with applied litter.

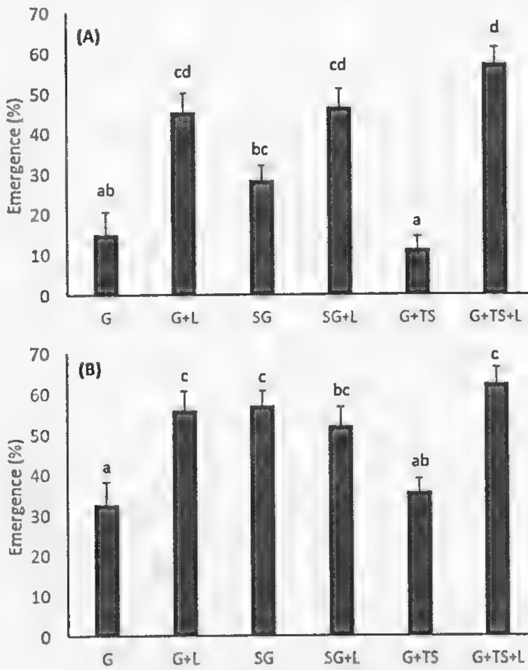

Figure 3. Emergence in the shade house of (**A**) *Acacia lamprocarpa* and (**B**) *Eucalyptus miniata* in gravel base (G), gravel base plus litter (G+L), scarified gravel (SG), scarified gravel plus litter (SG+L), gravel base plus topsoil (G+TS) and gravel base plus topsoil plus litter (G+ST+L). Bars that do not share the same letter above the bar are significantly different (GLM, quasibinomial with post hoc Tukey test, P<0.05).

Eucalyptus miniata showed a similar pattern of emergence in the shade house with over 50% emergence in the treatments where litter was applied over gravel, scarified gravel or topsoil. Scarifying the gravel alone resulted in similar levels of emergence to the litter treatments, and all these treatments had significantly higher emergence than from the compacted gravel base (33 ± 6%). Only topsoil application did not improve emergence over that of the compacted gravel base (the control).

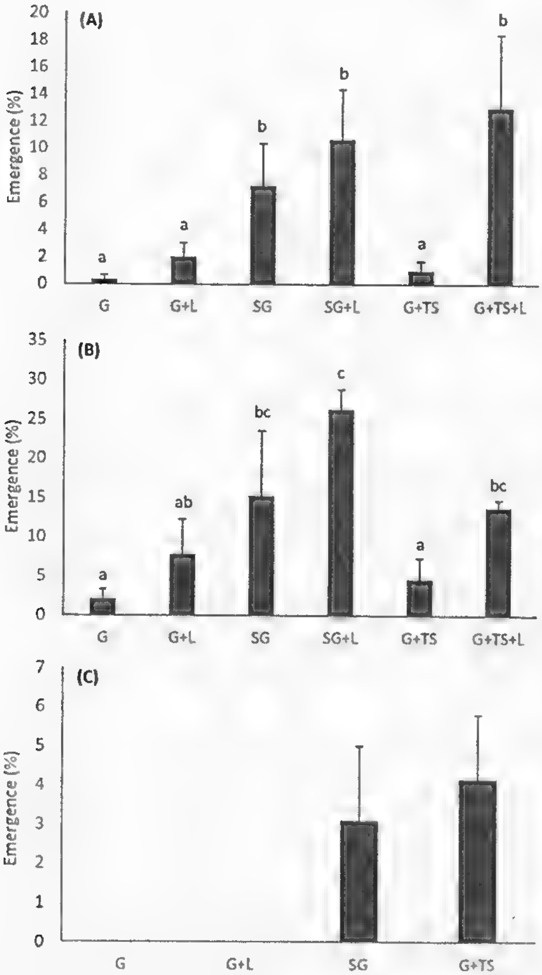

Figure 4. Emergence in the field plots of (**A**) *Acacia lamprocarpa*, (**B**) *Eucalyptus miniata* and (**C**) *E. tetrodonta* in gravel base (G), gravel base plus litter (G+L), scarified gravel (SG), scarified gravel plus litter (SG+L), gravel base plus topsoil (G+TS) and gravel base plus topsoil plus litter (G+ST+L). Bars that do not share the same letter above the bar are significantly different (GLM, binomial, with post hoc Tukey test, P<0.05).

Eucalyptus tetrodonta emergence from compact gravel in the shade house was 16 ± 8%.

In the field trial, mean emergence of all three species was much lower than that which occurred in the shade house. Emergence of *A. lamprocarpa* on compacted bare gravel was only 0.3 ± 0.3% (Figure 4). Emergence was significantly greater if the gravel base was scarified or if litter was applied. Application of 0.02 m depth of topsoil did not significantly increase emergence, unless litter was applied. One scarified gravel plot had 98% emergence of *Acacia* in comparison to the treatment mean of 7.3 ± 2.1%. This was excluded from statistical analysis as it was clearly an outlier.

Eucalyptus miniata emergence on bare gravel was only 2.0 ± 1.3%, whereas for *E. tetrodonta* no emergence on bare gravel occurred in the field (Figure 4). For *E. miniata,* the highest mean emergence of 17.3% resulted from the application of litter to scarified gravel, but scarifying the gravel or applying litter and topsoil also significantly increased emergence. Topsoil alone or litter applied to the gravel base did not significantly improve emergence. No treatments significantly improved emergence of

E. tetrodonta, although some emergence occurred in the scarified gravel plots and in the plots where soil was applied, and no emergence occurred in the untreated gravel.

Once the seedlings had emerged there were no significant effects of treatments on growth rates.

Mortality of emerged seedlings began within one month of emergence. Data on individual treatments for *E. tetrodonta* are not shown, as a total of only eight seedlings emerged. For *A. lamprocarpa* and *E. miniata,* mortality tended to be highest in the bare gravel and gravel plus litter plots (Table 1). Overall mortality tended to be lowest in the plots with scarified gravel or soil and litter. Scarified gravel had similarly low mortality for *A. lamprocarpa* but high mortality for *E. miniata.* Conversely, soil applied to gravel resulted in low mortality for *E. miniata* and high mortality for *A. lamprocarpa,* but the number of emerging seedlings was relatively low.

Table 1. Mortality of *Acacia lamprocarpa* and *Eucalyptus miniata* seedlings from March until May 2018 in the field trial. The number of emerging and surviving seedlings is the total for all plots.

	Acacia lamprocarpa			*Eucalyptus miniata*		
	Emerge #	Died #	Mortality %	Emerge #	Died #	Mortality %
Gravel	1	1	100%	5	2	40%
Gravel + Litter	6	3	50%	18	9	50%
Scarified Gravel	69	9	13%	33	12	36%
Scarified Gravel + Litter	33	5	15%	52	9	17%
Gravel + soil	3	2	67%	11	0	0%
Gravel + soil + litter	39	8	21%	45	7	16%

Discussion

For successful land restoration, research focused on the factors affecting establishment and finding cost-effective solutions is essential. As expected, surface treatments applied to the gravel base that remained after the gravel had been extracted affected emergence of *Acacia* and *Eucalyptus* seedlings. There was a dramatic increase in emergence with scarification of the gravel and the litter application treatments.

In this study of 700 seeds sown into the untreated gravel base, only six emerged and only three survived until May. It is quite likely that none of those 700 seeds would have produced seedlings that continued to survive throughout the first dry season. Recovery of vegetation in gravel extraction sites is very poor in the Darwin region (Price *et al.* 2005) and this study found very poor performance of seedlings of all studied species when sown on the untreated gravel base. Similarly, Setterfield *et al.* (1993) found that gravel borrow pits for road construction in Kakadu National Park recovered very little compared to their original vegetation. This may change with sufficient time, but is yet to be demonstrated.

There are several potential reasons for poor performance on the gravel substrate. There might be a physical hindrance to root growth (Rokich *et al.* 2001) or an inability to store sufficient water due to very low porosity (Ceacero *et al.* 2012). By hindering root growth, the compacted gravel substrate might also cause nutrient deficiencies in the seedlings. Hence, due to the physical properties of the gravel substrate, as well as prolonged lack of water and lack of nutrients in the abandoned gravel pit, young seedlings in the field might be more vulnerable to drought and have impaired anchorage, thus limiting their growth and survival.

We found that scarification of the gravel surface improved seedling emergence. Producing heterogeneity in the surface media, and creating microsites with more favourable conditions for seed germination and seedling establishment, can greatly assist tree establishment (Battaglia & Reid 1993). Setterfield (2002) found that seedling regeneration was limited by both seed supply and microsite availability in the Australian tropical savanna around Darwin. Scarification of the gravel surface affects water infiltration by creating hollows to trap water, which increases soil moisture. It also promotes litter accumulation in the hollows and the depressions may be shaded, which reduces temperature. Research done on seed germination and establishment of *Eucalyptus delegatensis* in different microsites (hillock, depression and flat site) in Tasmania showed that small scale variation in soil conditions even at the scale of tens of centimetres affected its germination and establishment (Battaglia & Reid 1993). Setterfield (2002) also found that shallow scarification of the soil surface in a tropical savanna increased seedling establishment of *Eucalyptus miniata* two to four times more than without disturbance. Likewise, seedling emergence of *Eucalyptus blakelyi* and *Acacia dealbata* was lower for seeds sown on untreated surfaces compared to seeds sown on scarified soil as seeds become shallowly buried and soil-to-seed contact is improved (Clarke & Davison 2001).

Litter improved the establishment success of the *Acacia* and *Eucalyptus* seedlings in our trials irrespective of topsoil application. Similarly, Facelli *et al.* (1999) observed higher densities of *Eucalyptus obliqua* seedlings in microsites with litter compared to those sites without it, due to enhancement of the humidity in the microenvironment surrounding the seeds. Litter favours seedling establishment in water-limited areas by reducing evaporative water loss from the soil, thus enhancing seed germination, growth and seedling survival (Enright & Lamont 1989). Removal of vegetation and topsoil from the gravel extraction site reduces the nutrient resources as most of the nutrient content is in the living biomass and the upper organic soil. Also, the moisture-holding capacity of gravel pits is reduced by loss of organic material and by surface compaction (Johnson 1987). Therefore, additional benefits for water and nutrition resources for plant growth can be expected as the litter breaks down. However, litter can be detrimental in some circumstances. If it is applied too thickly on top of the seeds, it may shade the seedlings, thus preventing photosynthesis, and the seedlings may die without emerging from the litter (Facelli *et al.* 1999). Seedling mortality in the shade house in our study was very low

(0–4%), but some mortality appeared to be due to litter promoting rotting of seedlings under moist conditions.

Due to weather conditions at the site, particularly occasional storms, loose leaf litter applied on the top of the gravel substrate was moved by wind and rain, and some plots were left with little or no litter at the end of the trial. For better results from litter application, the rehabilitation sites might be treated with leaf litter cut as foliage and small branches so that it cannot be easily moved by wind and rain.

An unexpected result was that a thin layer of topsoil was less favourable for seedling emergence than the application of a thin layer of leaf litter. During emergence, the topsoil is easily eroded by storms and the tiny root of the emerging seedling can be easily exposed and killed. An interesting observation was that although a 0.02 m depth of topsoil did not promote emergence, the 14 seedlings that emerged in topsoil over gravel had better overall survival than those growing on gravel. Once the seedling is able to establish in soil, the soil likely provides a better source of nutrients and water. However, in the Northern Territory, topsoil can also promote vigorous growth of grasses and numerous weed species, and sometimes competition can inhibit successful tree establishment and survival (Fawcett 1995). At our study site, clumps of grasses were only present in the piles of topsoil (although not in our treatment plots).

The regular supply of water in the shade house strongly influenced the germination, growth and survival of the seedlings. Seedling emergence was much greater in the shade house for all species (41%) than in the field (8%), suggesting that water stress was likely the major factor affecting survival in the field, however greater wind disturbance, seedling herbivory and erosion could have been additional factors. Similarly, Wilson & Bowman (1994) recorded 60–70% seed germination of *Eucalyptus miniata* and *E. tetrodonta* in nursery conditions compared to 3–15% in their field trial. In our trial, mortality of emerged seedlings was negligible in the shade house (0–4%) compared to the overall mortality rate of seedlings in field of 21%, indicating that mortality due to water stress and other factors is continuing to occur.

Seed availability is a major issue for restoration of gravel extraction sites. We did not observe any natural emergence of *Acacia* or *Eucalyptus* seedlings in our trial area outside our plots. When gravel extraction occurs, trees are cleared over substantial areas and seed dispersal of *Eucalyptus* species in particular is limited. They have no particular dispersal mechanisms and the seeds typically fall within a distance equivalent to the canopy height of the tree (Booth 2017). Over time, some *Eucalyptus* species of the savanna can also produce shoots from root buds, where lateral surface roots grow into the rehabilitation area. *Acacia lamprocarpa* does have a small yellow aril on its seed and so its seeds are adapted for ant dispersal (Dunlop *et al.* 1995). However, ants tend to remove seeds from disturbed areas and carry them into woodland rather than from the woodland into the open disturbed sites (Andersen & Morrison 1998). We also found that limited commercial seed supplies were a limiting factor for the trial.

For successful restoration of gravel extraction sites, research focused on finding solutions to factors affecting the revegetation success on those sites is necessary. The results of field experiments investigating different treatments and ways of improving soil conditions for plant growth can be used to substantially improve restoration outcomes. For example, scarification and application of litter can improve *Eucalyptus* establishment. Supplementary watering during the first dry season may also be an option at some sites but the effects of this on the developing seedlings would need to be investigated further.

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Across the Barkly: fauna recovery from a gas pipeline trench

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Abstract

This paper records the vertebrate fauna removed from a 481 km pipeline trench, which was the Northern Territory portion of a gas pipeline running from Tennant Creek, Northern Territory, to Mt Isa, Queensland. The project was conducted in two stages; the first from August to November 2017 and the second from May to July 2018. All the open section of the trench was checked daily during the entire construction of the pipeline, despite resourcing problems during the first stage. There were 14,613 animals retrieved comprising 113 species, with the majority (12,513 animals) recorded during the first stage. We compare our results with three previously published studies and discuss the problem of fauna mortality during pipeline construction.

Introduction

Pipelines for gas, water and other materials are constructed regularly throughout Australia. Pipes are placed into newly-dug trenches, and until the pipes are covered and buried, the trench can remain as a hazard for animals – open for days or even weeks. These pipeline projects traverse many hundreds of kilometres, and the open trench may extend for distances of just 50 m to more than 50 km at any one time. The trenches are, in effect, pitfall traps that require monitoring to reduce fauna fatalities.

There are several previous studies that provided inventories of animals recovered from pipeline trenches (Ayers & Wallace 1997; Faulkner 1999; Woinarski *et al* 2000; Doody *et al* 2003; Saipem 2008; Swan & Wilson 2012). The only inventories that covered the entire trench construction were the last three just mentioned. The others carried out selective sampling at various points during the construction.

Given the number of pipelines constructed in Australia over the last 30 years, we can only speculate as to why there have been so few published results on recovery of animals from the trenches. This is particularly significant considering the huge number of animals recorded on any project (often in the tens of thousands) and the number of secretive species that are seldom encountered using traditional sampling methods. The relevant state or territory fauna authorities should prioritise sufficiently rigorous monitoring processes to ensure that accurate lists of the fauna recovered are compiled

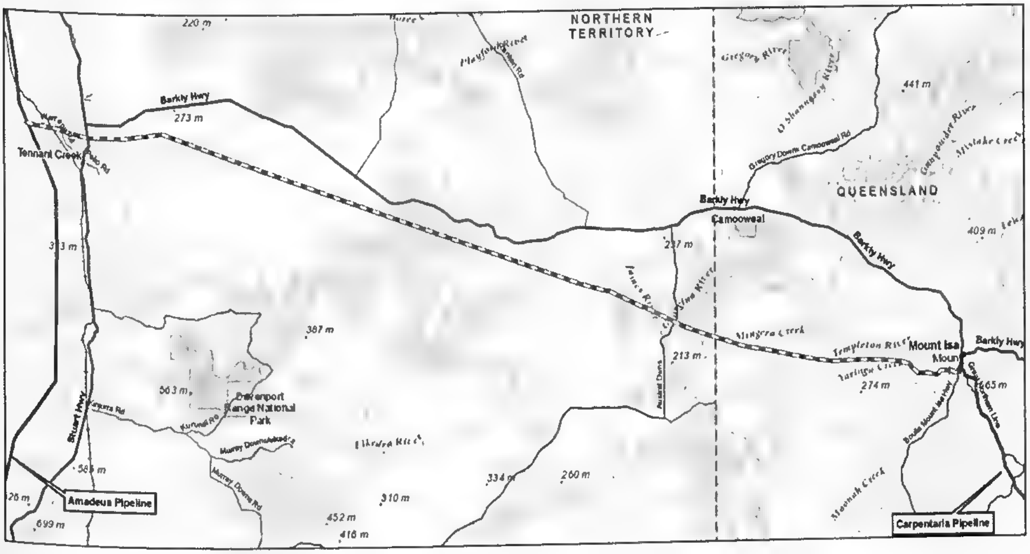

Figure 1. Route of the Northern Gas Pipeline.

and recorded into the relevant databases. There should also be standardised national operating procedures that help minimise impacts on fauna.

During 2017–2018 we were engaged by McConnell Dowell Constructors (Aust.) PL to carry out fauna recovery from the trench of the Northern Gas Pipeline which connects the gas fields in the Northern Territory to the Eastern Gas Market and runs from just west of Tennant Creek, Northern Territory, to Mount Isa, Queensland (Figure 1). The McConnell Dowell construction scope was for 481 km of 12 inch high-pressure gas pipeline to be split into two stages to avoid the monsoonal rains during summer. This region of the Northern Territory has two distinct seasons, wet and dry. The wet season occurs from October to April, however late December to February is when monsoonal rains, storms and high humidity generally occur. The majority of the annual rainfall occurs during this time with average rainfall for this region being 467.7 mm (BOM 2019).

Here we provide the results from the Northern Territory portion of the project, and compare our results with some of the earlier studies. The impact of seasonal variation on the fauna mortality during a pipeline project is also considered.

A small number of animals recorded in the data were rescued during clearing operations at the beginning of the project, and at the end when the stockpiled vegetation was spread back on the Right of Way.

Project site

The pipeline route traversed cattle grazing properties and Aboriginal land for the most part and did not encroach upon any national parks or conservation reserves.

During stage one, 262 km of trench was dug, had pipe lowered into it, and was backfilled over 86 days between August and November 2017. During this time maximum temperatures ranged between 25–40°C. This section of route traversed the Davenport-Murchison Ranges bioregion, sub-bioregions Ashburton and Barkly. The habitat consisted primarily of low, rugged rocky hills with hummock grasslands and low open eucalypt woodland dominated by *Acacia* species (IBRA 2012).

Fire had passed through about one quarter of the project site approximately one month before construction. Prior to this fire, mapping obtained from the Northern Australian Fire Information website (NAFI 2015) indicated that fire had burnt through approximately half the project footprint during 2003–2015. The most significant fire period was during 2011, when the majority of the first 354 km was burnt.

Stage two covered 219 km over 73 days during May to July 2018. The cooler months, provided far more favourable climatic conditions for successful fauna recovery, with maximum temperatures ranging between 21–32°C.

The landscape of stage two continued through the Davenport-Murchison Ranges bioregion and passed through the Tanami bioregion, sub-bioregion Sandover, for less than 50 km. This region is characterised by sand plains bisected by hills and rocky ranges topped with hummock grass (*Triodia*) and *Acacia* shrub-lands. The Sandover bioregion abruptly transitions into Mitchell Grass Downs, sub-bioregion Barkly Tablelands, which is characterised by cracking clay soils and Mitchell Grass (*Astrebla*) tussock grassland segmented by numerous river channels.

Methodology

A 30 m wide Right of Way was initially cleared to provide access for trench diggers and other machinery (Figure 2). The trench ranged in depth from 1.2–1.8 m. To assist the safe egress of the large and more mobile animals from the trench, fauna plugs, consisting of mounds of soil with slopes of 45 degrees, were positioned in the trench. It was expected that these were to be at least every 750 m. To provide cover and some protection from weather extremes two hessian sacks filled with sawdust were placed side by side against the wall of the trench every 200 m. The methodology of fauna recovery

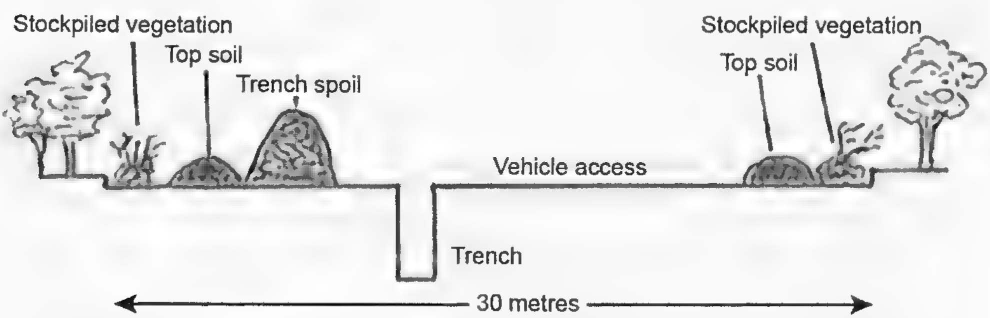

Figure 2. Schematic cross-section of the Right of Way either side of a pipeline trench.

from the open trench remained consistent throughout both stages of the project, although distances between fauna plugs varied and sometimes exceeded the expected 750 m distance. Daily fauna recovery involved walking the entire open trench to remove and relocate all vertebrates. Although it was a requirement for the trench check to be carried out within five hours of sunrise, this was not often achieved during stage one due to a lack of resources.

The trench inspection was conducted by two qualified fauna spotters/catchers walking each side of the trench, to provide a full view of the trench and prevent any fauna being missed. The sawdust bags were checked daily during the morning trench check, and additional sawdust bags were added to the new trench dug every afternoon. All live and dead vertebrates were identified to species level (using Wilson & Swan (2017), Van Dyck *et al.* (2013), Menkhorst & Knight (2011) and Tyler & Knight (2009)) and the details were recorded. All live animals were released in suitable microhabitats adjacent to their capture spot at 20–100 m from the trench. This information was entered daily into a database for McConnell Dowell. In many cases voucher images were taken (Figure 3).

To avoid working within the trench itself, the catchers used long-handled hooks and hoop bags to retrieve the animals. Slight modifications were made to the hoop bags used in the trench to recover the fauna for stage two. The hoops were designed specifically for the width of the trench, which prevented fauna from running past the bag, and thus providing more efficient capture of, and less stress to, the animals.

Results

Table 1 summarises the species and numbers caught on each of the two stages.

In total, we removed 113 vertebrate species from the open trench during the laying of the pipe. Reptile species were the most common, comprising 84 species (75%), then mammals (18 species, 16%), then frogs (5 species, 4%) and then birds (6 species, 5%). Exotic animals removed from the trench were mice, cattle and one dingo. We did not record the locations of these exotic species and few feral animals were encountered, although cat and dog footprints were commonly seen.

It must also be assumed that some animals fell into the trench on more than one occasion. Without marking each individual it was impossible to tell the extent to which this might have happened. However, any particular section of trench was usually not open for more than a few days, and to minimise re-entry, animals were relocated into appropriate refuges up to 100 m from the trench, and where possible on the spoil-side.

Of the 14,613 animals recovered, 13,838 (94.7%) were alive and 775 (5.3%) were dead. Some were killed during the trenching operation itself, but most died from exposure to the sun or predation by other animals, including cats and dingos. We were continually surprised at the high numbers of very small animals found in the trench. Considering the obstacles presented by the Right of Way (Figure 2), it is a wonder that any animals at all got as far as the trench!

The most frequently recorded species was the Northern Spiny-tailed Gecko (*Strophurus ciliaris*) (4120 individuals encountered). This was followed by the Central Netted Dragon (*Ctenophorus nuchalis*) (2554 individuals). Other than the Dusky Ctenotus (*Ctenotus helenae*) (521 individuals) and the Northern Narrow-banded Skink (*Eremiascincus intermedius*) (217 individuals), skinks were not caught in very large numbers and it may be that some of the fossorial species avoided detection in loose spoil on the floor of the trench.

There were 735 snakes removed with the most frequently recorded species being Little Spotted Snake (*Suta punctata*) (214 individuals). The next most commonly encountered snake was the Curl Snake (*Suta suta*) (114 individuals).

Among the mammals, most occurred in low numbers except for the Spinifex Hopping Mouse (*Notomys alexis*) (699 individuals) and the Fat-tailed Dunnart (*Sminthopsis crassicaudata*) (295 individuals).

Frogs were scarce in contrast to previous projects where they were often the most numerous taxa to be recovered from the trench. Consistent and heavy rain would certainly have brought them to the surface. Most found were burrowing frogs, some of which had been dug out by the trenching machines and were often still encased in their membranous cocoons. On one occasion, over two days during a hot dry period, a number of *Cyclorana* specimens were encountered active on the Right of Way and in the trench itself, still partly encased in membranes. It is suspected that the noise or vibrations of the heavy machinery mimicked the stimulus of heavy rains, which instigated their emergence to the surface.

Thirteen birds were found and these all appeared to be individuals that found their way in but were unable to fly out of the narrow trench.

Figure 3 (on oposite page). Reptile species recorded along the Northern Gas Pipeline.

a. No Shine's Whipsnakes (*Demansia shinei*) were seen active along the pipeline route, yet 44 were recorded during construction. (Steve Wilson)

b. The Northern Spiny-tailed Gecko (*Strophurus ciliaris*) was by far the most frequently encountered vertebrate. (Steve Wilson)

c. Just one Eastern Striped Ctenotus (*Ctenotus robustus*) was recorded on the project. (Steve Wilson)

d. The Pale-backed Ctenotus (*Ctenotus pallescens*) is a poorly known species. There were 53 specimens recorded, illustrating the effectiveness of pipeline monitoring for revealing cryptic animals. (Steve Wilson)

e. The Pretty Ctenotus (*Ctenotus pulchellus*) was found at the eastern end of the pipeline, in Mitchell Grass Downs during stage two. (Christy Harvey)

f. The Centralian Earless Dragon (*Tympanocryptis* cf. *centralis*) was only recorded on the western end of the pipeline, on loamy and stony soils near Tennant Creek. (Steve Wilson)

g. For some predators, a pipeline trench is a feeding opportunity. This Sand Goanna (*Varanus gouldii*) is swallowing a Central Netted Dragon (*Ctenophorus nuchalis*). (Steve Wilson)

h. The Desert Pygmy Monitor (*Varanus eremius*) is a secretive inhabitant of arid areas with *Triodia*. Though none were seen active, 32 were removed from pipeline trenches or uncovered during clearing operations. (Steve Wilson)

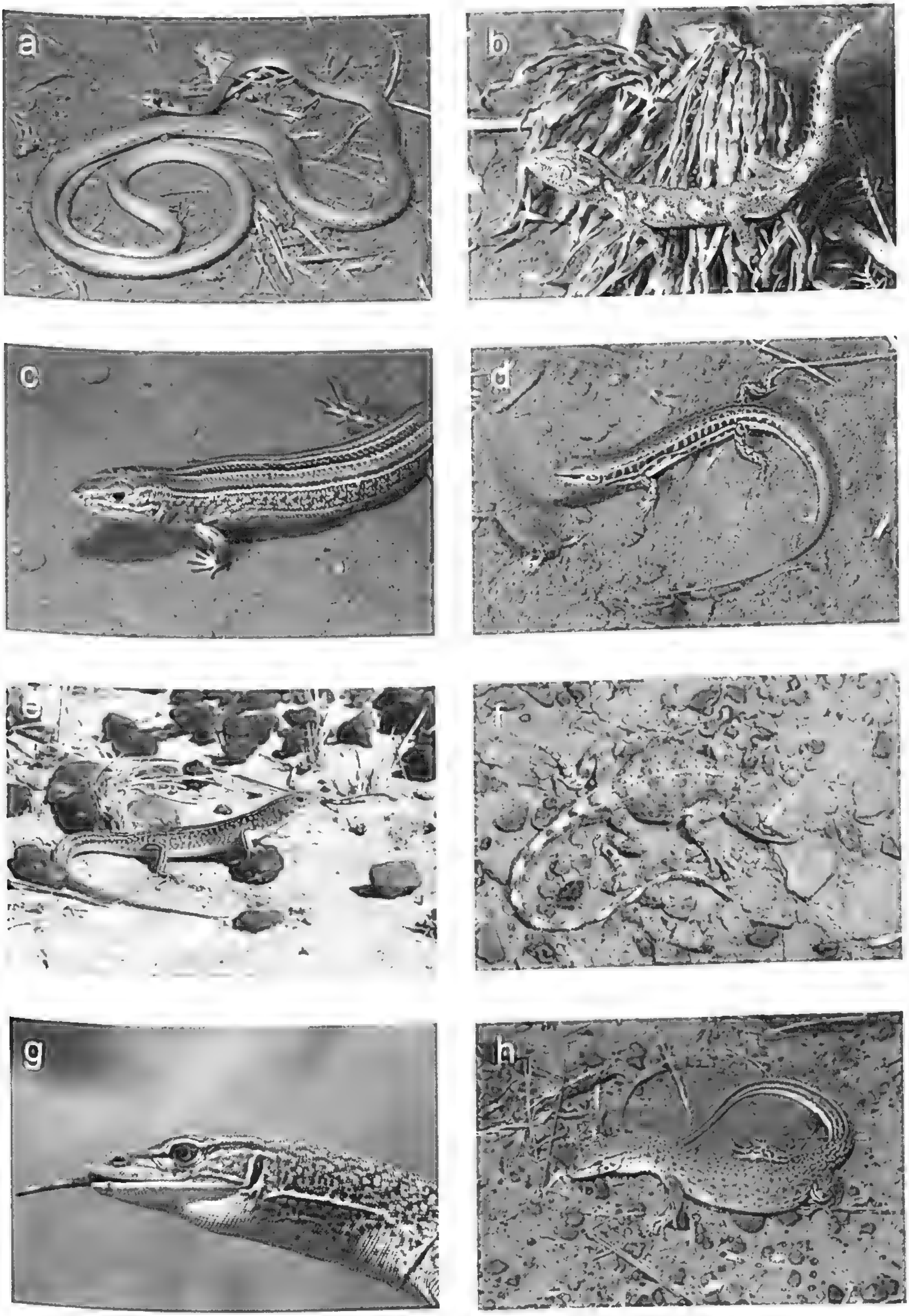
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Table 1. List of fauna removed from the pipeline trench during the two cycles.

Species	Common name	2017	2018	Total
Dasyurid marsupials				
Dasycercus blythi	Brush-tailed Mulgara	1	7	8
Planigale ingrami	Long-tailed Planigale		43	43
Planigale tenuirostris	Narrow-nosed Planigale	2		2
Ningaui ridei	Wongai Ningaui	5		5
Sminthopsis crassicaudata	Fat-tailed Dunnart	294	1	295
Sminthopsis macroura	Stripe-faced Dunnart	4	77	81
Sminthopsis ooldea	Ooldea Dunnart	62		62
Sminthopsis sp.	Dunnart	1		1
Sminthopsis youngsoni	Lesser Hairy-footed Dunnart	2	14	16
Macropods				
Macropus rufus	Red Kangaroo	2		2
Rodents				
Leggadina forresti	Central Short-tailed Mouse	13	30	43
Notomys alexis	Spinifex Hopping Mouse	577	122	699
Notomys sp.	Hopping Mouse	1		1
Pseudomys delicatulus	Delicate Mouse	34	20	54
Pseudomys desertor	Desert Mouse	15	1	16
Pseudomys hermannsburgensis	Sandy Inland Mouse	175	10	185
Pseudomys johnsoni	Central Pebble-mound Mouse		1	1
Feral/stock				
Mus musculus	House Mouse	1		1
Canis lupus dingo	Dingo	2		2
Bos indicus	Cattle	3		3
Lizards – geckos				
Nephrurus levis	Smooth Knob-tailed Gecko	3		3
Amalosia rhombifer	Zigzag Velvet Gecko	10		10
Diplodactylus conspicillatus	Variable Fat-tailed Gecko	38	15	53
Diplodactylus laevis	Desert Fat-tailed Gecko	741	12	753
Diplodactylus tessellatus	Tessellated Gecko	2	18	20
Lucasium immaculatum	Pale-striped Ground Gecko	22		22
Lucasium stenodactylus	Sand-plain Gecko	381	11	392
Rhynchoedura ornata	Western Beaked Gecko	521	50	571
Strophurus ciliaris	Northern Spiny-tailed Gecko	3306	814	4120
Strophurus jeanae	Southern Phasmid Gecko	127	4	131
Gehyra minuta	Dwarf Dtella	3		3
Gehyra pilbara	Pilbara Dtella	1		1
Gehyra purpurascens	Purple Dtella	378	11	389
Gehyra versicolor	Variable Dtella	32	2	34
Heteronotia binoei	Bynoe's Gecko	181	28	209
Lizards – flap-footed				
Delma borea	Northern Delma	46	5	51
Delma butleri	Spinifex Delma	8	6	14

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Species	Common name	2017	2018	Total
Delma cf. *haroldi*	Neck-barred Delma	1		1
Delma tincta	Black-necked Delma	2		2
Lialis burtonis	Burton's Snake-lizard	103	3	106
Pygopus nigriceps	Western Hooded Scaly-foot	32	1	33
Pygopus steelescotti	Northern Hooded Scaly-foot		1	1
Lizards – skinks				
Carlia munda	Striped Rainbow Skink	18		18
Carlia triacantha	Desert Rainbow Skink	8		8
Carlia sp.	Rainbow Skink		3	3
Cryptoblepharus australis	Inland Snake-eyed Skink	1	1	2
Cryptoblepharus metallicus	Metallic Snake-eyed Skink	7		7
Ctenotus grandis	Giant Desert Ctenotus	2		2
Ctenotus greeri	Greer's Ctenotus	139	10	149
Ctenotus helenae	Dusky Ctenotus	490	31	521
Ctenotus joanae	Blacksoil Ctenotus		1	1
Ctenotus leonhardii	Common Desert Ctenotus	119	13	132
Ctenotus pallescens	Pale-backed Ctenotus	24	29	53
Ctenotus cf. *pallidus*	Ctenotus	2		2
Ctenotus pantherinus	Leopard Ctenotus	13	41	54
Ctenotus piankai	Pianka's Ctenotus	17	8	25
Ctenotus pulchellus	Pretty Ctenotus		2	2
Ctenotus robustus	Eastern Striped Ctenotus	1		1
Ctenotus saxatilis	Rock Ctenotus	37		37
Ctenotus schomburgkii	Barred Wedge-snouted Ctenotus	14	3	17
Ctenotus sp.	Ctenotus	1		1
Eremiascincus intermedius	Northern Narrow-banded Skink	210	7	217
Lerista aericeps	Yellow-tailed Slider	1	1	2
Lerista bipes	Western Two-toed Slider	42	54	96
Lerista sp.	Skink	3		3
Menetia greyii	Common Dwarf Skink	13	1	14
Morethia ruficauda	Fire-tailed Skink	3		3
Notoscincus ornatus	Ornate Snake-eyed Skink	37	12	49
Proablepharus kinghorni	Kinghorn's Snake-eyed Skink		1	1
Proablepharus reginae	Spinifex Snake-eyed Skink	17		17
Tiliqua multifasciata	Centralian Blue-tongue	112	12	124
Lizards – agamids				
Amphibolurus centralis	Centralian tree Dragon	18		18
Ctenophorus isolepis	Central Military Dragon	352	53	405
Ctenophorus nuchalis	Central Netted Dragon	2411	143	2554
Diporiphora lalliae	Lally's Two-lined Dragon	247	13	260
Diporiphora cf winneckei	Canegrass Dragon		7	7
Diporiphora sp.	Dragon	3		3
Gowidon longirostris	Long-nosed Dragon	100	60	160

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Species	Common name	2017	2018	Total
Moloch horridus	Thorny Devil	14	9	23
Pogona minor	Dwarf Bearded Dragon	107	21	128
Pogona vitticeps	Central Bearded Dragon	2	9	11
Tympanocryptis cf. centralis	Centralian Earless Dragon	11		11
Tympanocryptis tetraporophora	Eyrean Earless Dragon		4	4
Lizards – monitors				
Varanus acanthurus	Spiny-tailed Monitor	35	1	36
Varanus brevicauda	Short-tailed Pygmy Monitor	1	1	2
Varanus eremius	Pygmy Desert Monitor	28	4	32
Varanus gilleni	Pygmy Mulga Monitor	32	4	36
Varanus gouldii	Sand Goanna	72	3	75
Varanus panoptes	Yellow-spotted Monitor	3		3
Varanus tristis	Black-headed Monitor	10		10
Snakes – blind				
Anilios diversus	Northern Blind Snake	85	3	88
Anilios dibersus cf. *grypus*	Blind Snake	1		1
Anilios endoterus	Desert Blind Snake	5	1	6
Anilios sp.	Blind Snake	2		2
Snakes – pythons				
Antaresia stimsoni	Stimson's Python	6	9	15
Aspidites melanocephalus	Black-headed Python	7	4	11
Aspidites ramsayi	Woma	8	2	10
Snakes - elapids				
Brachyurophis incinctus	Unbanded Shovel-nosed Snake	1		1
Brachyurophis roperi	Northern Shovel-nosed Snake	57		57
Demansia rimicola	Blacksoil Whipsnake		3	3
Demansia shinei	Shine's Whipsnake	39	5	44
Furina ornata	Orange-naped Snake	18		18
Pseudechis australis	Mulga Snake	24	1	25
Pseudonaja guttata	Speckled Brown Snake		5	5
Pseudonaja mengdeni	Western Brown Snake	56	4	60
Pseudonaja modesta	Ringed Brown Snake	51	8	59
Pseudonaja textilis	Common Brown Snake	2		2
Suta punctata	Little Spotted Snake	181	33	214
Suta suta	Curl Snake		114	114
Frogs – tree				
Cyclorana australis	Giant Burrowing Frog		27	27
Cyclorana cultripes	Knife-footed Frog	24		24
Cyclorana longipes	Long-footed Frog	1		1
Cyclorana sp.	Burrowing Frog	8		8
Litoria rubella	Desert Tree Frog		9	9
Frogs – Australian ground				
Neobatrachus sp.	Frog	7		7

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Species	Common name	2017	2018	Total
Notaden nichollsi	Desert Spadefoot	1		1
Unidentified frogs		10		10
Birds				
Anas gracilis	Grey Teal	1		1
Coturnix coturnix	Common Quail	6		6
Falco berigora	Brown Falcon	1		1
Lichmea indistincta	Brown Honeyeater	1		1
Tachybaptus novaehollandiae	Australasian Grebe	1		1
Turnix velox	Painted Buttonquail		3	3

Table 2. Summary of fauna removed from the trench.

Taxon	Number of individuals	Number of species
Mammals	1514	18
Lizards	12,258	66
Snakes	735	18
Frogs	87	5
Birds	13	6
Total	14,607*	113

* Total number of individuals excludes six feral/livestock.

Discussion

Stage one of the pipeline began about 40 km west of Tennant Creek at KP (Kilometre Point) 0 in August 2017 and concluded at KP262 in November that year, so it was in operation throughout the spring months. It largely traversed hummock grass sandplains. This is reflected in the high number of animals caught (12,513). It was common for more than 100 animals to be removed from the trench per day. A spike in humidity over several days during late October resulted in maximum vertebrate activity with 644 animals removed on 29 October; 530 on 30 October and 563 on 31 October. In comparison, stage two commenced at KP262, to the east of Tennant Creek during winter (May to July) and only 2100 animals were recorded. At first it continued through hummock grass sandplains, but then progressed through Mitchell Grass Downs with cracking black soil.

Changes in habitat were clearly reflected by the presence or apparent absence of a number of species. *Suta punctata* was numerous throughout the hummock grass areas but was replaced by *Suta suta* as soon as construction transitioned into the Mitchell Grass habitat. *Tympanocryptis* cf. *centralis, Lucasium immaculatum* and *Amphibolurus centralis* were recorded at the very start of the project at the Tennant Creek end, but they cut out or declined eastwards. The 11 *T.* cf. *centralis* were found only between KP0 and KP21. Of 22 *L. immaculatum,* 20 were located between KP6 and KP49, while just 2 were recorded further east at KP103 and KP107. Likewise, of 18 *A. centralis* recorded, 16 were found between KP6 and KP35, and just two further east at KP111.The sharp drop in these species is despite warmer, more favourable conditions. It may be related to substrate,

as most were on heavier loams and stony soils rather than the vast sandplains further east. *Pogona minor* was a commonly recorded species on the loams and sandplains, and probably reaches its limit on the western edge of the Mitchell Grass Downs, approx. 145 km west of the Queensland/Northern Territory border.

Monitoring of pipeline trenches reveals cryptic animals that are potentially common, but rarely seen by other means. Shine's Whipsnake (*Demansia shinei)* is one such example. Forty four were seen, all in trenches except one, which was located in the grounds of a Tennant Creek motel. None were seen active on the ground in spite of the thousands of kilometres driven on roads and tracks and hundreds of kilometres walked through suitable habitat at all times of day.

We only recorded one individual of the Eastern Striped Ctenotus (*Ctenotus robustus*). This was taxonomically significant as the type locality, Barrow Creek, lies 223 km to the south and there has been some doubt expressed regarding the validity of type data. This record confirms *C. robustus* as a Central Australian species.

The gecko *Strophurus ciliaris* was the big surprise. The total number recorded was 4120, which was far in excess of any other lizard, and the most abundant vertebrate recorded. It was not uncommon to look along the trench to see seven or eight individuals scattered along the trench floor. This species of gecko is clearly a very significant element of the Barkly fauna.

Predation in the trench occurred quite regularly as the remains of victims were often encountered and we did observe instances of animals being consumed. The predators observed were the snakes and goannas, but we also saw many footprints of cats and dogs. Presumably they were also having a feast.

The pythons *Aspidites ramsayi* and *A. melanocephalus* were both recorded in the trench and along access tracks (see following paper in this issue of *Northern Territory Naturalist*). We found *A. melanocephalus* in the Tennant Creek area, which would appear on the south-western edge of its distribution. *Aspidites ramsayi* was first recorded at KP130, and both species continued to be recorded until the hummock grasses were replaced by Mitchell Grass Downs.

Fauna recovery on pipeline projects has the purpose of minimising the impact of the construction on native vertebrates by speedy removal from the trench with minimum handling, and relocating into suitable microhabitat.

There were mixed results on this project. Overall there were 13,838 live animals removed (94.7%) and 775 dead animals (5.3%). Stage one recovered 11,756 animals alive (94%) and 757 dead (6%). Stage two recovered 2082 animals alive (99.1%) and 18 (0.9%) dead.

Stage one was under-resourced in terms of suitably qualified and experienced people and this, coupled with the volume of animals, weather extremes and difficulty of access to the Right of Way meant that it was often not possible to check and clear the trench, relocate animals and record data within 5 hours of sunrise as expected. There was also

the problem of some animals, particularly dragon lizards, entering the trench after it had been checked. Due to latitude and time of year there was a period of several hours each day from late October into November when the angle of the sun eliminated all shade within the trench. Efforts were further complicated by gaps between fauna ramps frequently exceeding proscribed limits and the premature removal of fauna ramps.

Stage two was more adequately resourced and, with milder temperatures and fewer animals, the work was able to be completed within the designated timeframe. The trench was also not exposed to full sun at any time during this stage.

Table 3 compares this project with three others in terms of numbers recorded. In three of the four projects, reptiles were the most numerous fauna component. As the average number of species recorded per kilometre increased, so did the mortality.

Table 3. Comparison with previous studies. Figures within brackets are number of species.

	Km	Reptiles	Frogs	Mammals	Birds	Total No.	Average No per Km	Mortality (%)
Doody *et al.* (2002) Eastern Gas P'line	792	2960 (45)	4103 (19)	207 (17)	31 (14)	7301	9.22	3.06
Swan & Wilson (2012) Nth Qld Gas P'line	392	1631 (56)	1528 (18)	139 (14)	3 (3)	3301	8.42	0.85
Saipem (2008) DBNGP Stage 5a	571	9642	81*	2558		12,281	21.5	4.42
This study NGP	460	12,993 (84)	87 (5)	1514 (18)	13 (6)	14,607**	31.8	5.3

* The Saipem report showed 81 'other' which we assumed to be frogs.
** Total number of individuals excludes six feral/livestock.

There is a need, before any pipeline project commences, to set up a committee involving representatives of the pipeline company, contractors, fauna catchers and agencies for the exchange of information on clearing and trenching plans and progress, resource requirements for fauna work and changes to environmental management plans. There should also be serious consideration regarding the timing of a project, as this study clearly demonstrates a much lower fauna impact when construction occurs during cooler months.

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An observation of excavating behaviour by a Black-headed Python (*Aspidites melanocephalus*) in the wild

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Abstract

The Black-headed Python (*Aspidites melanocephalus*) and the Woma (*Aspidites ramsayi*) have both been reported as carrying out burrowing or excavating behaviour. These reports have been based mainly on observations of captive individuals, with the only observations of specimens in the wild being those of Bruton (2013) on Womas. Here we report on a Black-headed Python scooping out sand with its head and fore-body to create a depression in the wild.

The pythonid genus *Aspidites* has been reported as exhibiting burrowing behaviour (Ross & Marzec 1990; Ehmann 1993; Barker & Barker 1994), based mainly on the report by Murphy, Lamoreaux & Barker (1981) that four captive Black-headed Pythons (*A. melanocephalus*) excavated gravel by using their head and neck to scoop loose material and create a cavity. O'Brien & Naylor (1987) reported that a young specimen that had been recently removed from the wild and was being held pending release, was observed digging beneath rocks and logs, ultimately creating a cavity in which it concealed itself.

Fyfe & Harvey (1981) recorded similar behaviour by six captive Womas (*Aspidites ramsayi*). The floor of the vivaria in which they were housed was covered with 5–15 cm of sand and the pythons scooped this out in large quantities until they reached the base of the vivarium. Two of these Womas also refused to shelter in the hollow logs or small bushes provided. Instead, they rested on top of the sand with the front 50 cm of the body including the head submerged under the sand. They would remain in this position for up to 20 minutes before surfacing for a few minutes then repeating the movement. The only record of excavation in the wild was made by Bruton (2013), who reported two instances of wild Womas excavating burrows. In both cases, it appeared that they were enlarging existing burrows either for shelter or during hunting.

During survey work on the Northern Gas Pipeline project southwest of the Barkly Tableland in the Northern Territory (19°54.461'S, 135°56.692'E) we observed a Black-headed Python excavating soft sand in the bottom of a pipeline trench. The observation was made on 8 November 2017 during the daily check of the trench for animals that had fallen into it (see also preceding paper in this issue of *Northern Territory Naturalist*). We had

seen the track of a python on the floor of the trench, heading in the same direction we were. This continued for almost a kilometre, until we located the animal, approximately 1.3 m in length, scooping into loose sand with its head and neck (Figure 1), thus creating a shallow depression. It had only just begun this activity when we arrived and continued to scoop into the sand while we watched. At the time of day (10.45 hr) the temperature was around 35^{0} C and full sunlight was just beginning to flood the bottom of the trench, thus eliminating any shade. The depth of the trench was approximately 1.8 m with steep unstable edges, preventing the python from scaling the wall. We observed this digging behaviour for several minutes before we removed the animal from the trench and relocated it a suitable distance into adjacent low sparse *Eucalyptus* woodland with hummock grass.

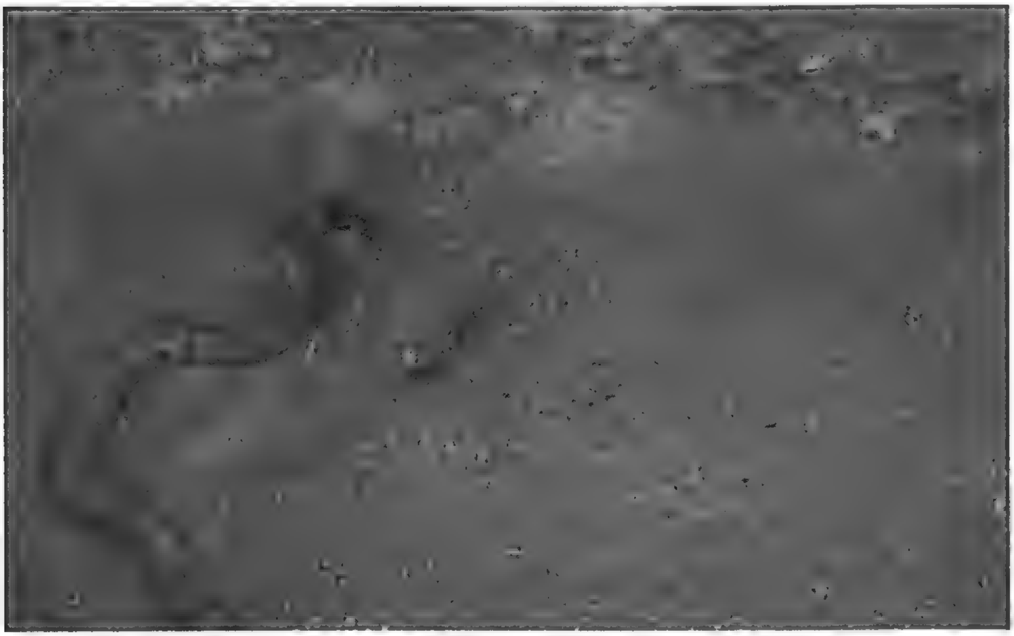

Figure 1. Black-headed Python excavating a depression into loose dry sand, southwest of Barkly Tableland, Northern Territory, 8 November 2017. (Gerry Swan)

In the case reported here, the sand was dry and very loose, and quite unsuitable for the creation of a burrow. Given the high temperature and the aspect of the sun in the trench we surmise that the python was attempting to create a depression sufficient to cover itself and escape the direct sunlight.

While some observations appear to be excavations for concealment or to uncover prey items, the purpose of others is not so clear. Unfortunately observations on captive behaviour do not necessarily reflect what happens in the wild. In this particular observation, while the python was not in captivity it was not in its natural environment, and what effect the confinement to a narrow trench would have is uncertain. However, the animal did engage in an excavation behaviour for a purpose that remains unclear.

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An assessment of thermal preference of two species of Knob-tailed Geckos, *Nephrurus levis* and *N. laevissimus*, at Uluru Kata-Tjuta National Park

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Abstract

Nocturnal lizards are often active at temperatures that are sub-optimal for physiological functioning, a phenomenon that has been referred to as the 'nocturnal paradox'. The purpose of this study was to investigate the general habitat and thermal preference of two species of nocturnal gecko in the genus *Nephrurus* with a focus on differences between preferred body temperatures determined under laboratory conditions and those measured in animals at the time of field collection. The Smooth Knob-tailed gecko (*Nephrurus levis*) and the Pale Knob-tailed Gecko (*N. laevissimus*) inhabit the desert environment of Uluru Kata-Tjuta National Park in the Northern Territory of Australia. Habitat preferences were determined by documenting capture locations for these species while thermal preferences were determined using laboratory-based thermal gradients. Analysis of habitat use demonstrated a significant difference in habitat preference between the two species. *Nephrurus levis* was most often in *Spinifex* sandplain and mulga shrubland and *N. laevissimus* was most often found around sand dune habitats. Field-active body temperatures of both species ranged from 14.5 °C to 32.2 °C and were significantly correlated with air and ground temperature at the time of capture. The thermal preference (T*pref*) of *N. levis* (mean day $Tpref_{mid}$ = 29.1 °C and mean night $Tpref_{24}$ = 28.2 °C, n = 19) and *N. laevissimus* (mean day $Tpref_{mid}$ = 29.5 °C and mean night $Tpref_{24}$ = 27.8 °C, n = 27) were not significantly different, although both species exhibited significantly higher daytime and night-time preferred body temperatures than body temperatures observed in the field. Body size did not affect thermal preference for either species. As such, the thermal preferences of these species support the concept of the nocturnal paradox. Additionally, it has been suggested that the characteristic swollen tail tip displayed by all *Nephrurus* species may play a role in assessing the thermal environment. While not investigated extensively here, combined data for both species demonstrated that individuals oriented their tail toward the heat source in thermal gradients significantly more than expected if orientation were random.

Introduction

Nocturnality in reptiles has been called a paradox since the organisms are commonly active at temperatures that are sub-optimal for traits such as sustained locomotion, sprint speed, growth and possibly auditory sensitivity (Campbell 1969; Huey *et al.* 1989; Autumn & DeNardo 1995; Autumn *et al.* 1999). For example, preferred body temperatures (T*pref*) in nocturnal geckos, determined by measuring the body temperatures selected by lizards in a laboratory thermal gradient, have been found to range between 25 °C to 35 °C (Dawson 1975; Angilletta & Werner 1998; Huey *et al.* 1989; Angilletta *et al.* 1999), which is higher than active nocturnal foraging body temperatures that have been recorded as low as 15 °C in some species (Meiri *et al.* 2013). Nocturnality in lizards may therefore represent a trade-off between physiological performance and access to an unexploited niche (Vitt *et al.* 2003). In the specific case of geckos, termites may have been a driver for the evolution of nocturnality since termites and their invertebrate predators are often active above ground at night (Morton & James 1988). Additionally, as termites occur in colonies, such a concentrated food supply assures that the profits gained from this resource will generally outweigh the costs of finding it (MacArthur & Pianka 1966; Pianka & Pianka 1976). Activity at night may also facilitate predator avoidance, as many lizard predators (e.g. birds) are either inactive at night or have decreased capacity to see prey. Additionally, studies have shown that nocturnal lizards have a lower cost of locomotion than their diurnal counterparts (Autumn *et al.* 1994, 1997, 1999), which may impart additional advantages.

One genus of gecko that has received limited attention in thermal studies is *Nephrurus*, the knob-tailed gecko. Pianka & Pianka (1976) reported that the field active body temperature (T*b*) of 12 nocturnal lizard species in Western Australia, including the Smooth Knob-tailed Gecko (*N. levis*), the Pale Knob-tailed Gecko (*Nephrurus laevissimus*), and the Midline knob-tailed Gecko (*N. vertebralis*), were strongly correlated with ambient air temperatures (T*a*). These data suggest that these species may be predominately thermoconforming to environmental temperatures during activity. The only published T*pref* data for *Nephrurus* is by Angilletta & Werner (1998), who recorded a mean T*pref* of 25.1 °C for the Starred Knob-tailed Gecko (*N. stellatus*). Such a gap in T*pref* data for *Nephrurus* geckos hinders the interpretation of T*b* data. Additionally, a notable and unique feature of this genus is the swollen knob at the tip of the tail. The function of the caudal knob is still largely unknown, however some have suggested it may play a role in assessing the thermal environment (Russell & Bauer 1987).

Two species of *Nephrurus*, *N. levis* (Figure 1) and *N. laevissimus* (Figure 2), inhabit the central Australian deserts, where they are relatively abundant and can be found foraging at night on the open sand of dunes and sandplains. Although these two species are similar in morphology, they have been reported occupying different microhabitats within the Australian landscape. Studies have found that *N. levis* appears to prefer sandplain habitats, at least in the western half of its range, whilst *N. laevissimus* generally occupies sand dune habitats (Pianka 1969; Pianka 1972; Pianka & Pianka 1976). The

Figure 1. Smooth Knob-tailed Gecko (*Nephrurus levis*). Uluru Kata-Tjuta National Park. Captive individual in thermal gradient. (Brenton Hays)

Figure 2. Pale Knob-tailed Gecko (*Nephrurus laevissimus*). Uluru Kata-Tjuta National Park. Captive individual in thermal gradient. (Brenton Hays)

environment and ecology of the two species offers an ideal framework for an investigation of how temperatures selected in a thermal gradient, i.e. thermal preference (T*pref*) compare to field-active body temperatures (T*b*) experienced at night. This study had two aims; 1) to measure temperature selection (T*pref*) for *N. levis* and *N. laevissimus* in a laboratory thermal gradient and 2) to compare T*pref* with T*b* both within and between the two species. As a secondary objective, this study aimed to collect preliminary data regarding the role of the tail knob in the animal's assessment of the thermal environment.

Materials and Methods

Study site

Geckos were sampled from Uluru Kata-Tjuta National Park between September and November 2013 during peak activity times for the study species. The Park covers an area of 1325 km^2 and contains two major rock formations: Uluru and Kata-Tjuta. The climate in the area is arid, with low rainfall (generally occurring in summer) and high evaporation (Reid *et al.* 1993). Mean daily minimum and maximum air temperatures range from 20.9 °C to 37.5 °C in January (mid-summer) and 3.4 °C to 20.3 °C in July (mid-winter) (Australian Government Bureau of Meteorology 2014).

Field collection

Surveys typically commenced at approx. 20.00 hr and continued for an average of four hours. Survey sites were chosen based on known *Nephrurus* habitat (Pianka & Pianka 1976; Reid *et al.* 1993), including sand dunes (crest and swale), *Spinifex* sandplains, and Mulga shrublands. Using a head torch and hand-held flashlight, geckos were detected by movement or eye shine. At times geckos were also sampled via vehicle cruising on the bitumen roads and dirt tracks. Once detected, geckos were captured by hand

as quickly as possible. Once captured, body temperature was measured within 15 sec of handling (Mayhew 1963) by inserting the thermocouple of a Fluke® 566 infrared thermometer (Fluke Australia Pty Ltd, Castle Hill, NSW) in the cloaca. This temperature was recorded as the gecko's field-active body temperature (T*b*). Air (T*a*) and ground (T*g*) temperature were also recorded at the site of capture. Air temperature was measured by placing the Fluke® thermocouple approximately 5 cm above the ground surface. The infrared thermometer was also used to measure ground surface temperature. Habitat type where geckos were sampled was noted as sand dune, *Spinifex* sandplain, or mulga shrubland. For the purpose of this study, sand dune habitats included the swale, which was classified as the area surrounding the dune to a distance of approximately 25 m, depending on both the distance to neighbouring dunes and the slope of the ground (i.e. the base of some dunes were more obvious than others). Identification of habitat types followed descriptions by Pianka (1972), Buckley (1981) and Saxon (1984). Geckos were then placed in marked cloth bags and stored temporarily in an ice-chilled (approx. 10 °C) and insulated cooler for transportation back to the laboratory (Klawinski *et al.* 1994). Sites where geckos were found were not revisited to reduce the likelihood of double sampling.

Measurement of thermal preference

The laboratory in which the thermal preference experiments were conducted was a 10 x 3 m demountable building located within the national park. Thermal preference was measured using thigmothermic thermal gradients based on design specifications used by Angilletta & Werner (1998). Thermal gradients were constructed from 12 mm plywood for the sides and 2 mm galvanised steel for the base. Fine sand sampled from the top of a sand dune was spread approximately 5 mm across the metal base to provide substrate. Each gradient was 1200 x 600 x 200 mm and consisted of three separate runs. Three of these gradients were constructed, accommodating nine geckos at any one time. A temperature gradient was created using 120-watt heat lamps for the heat source (one heat lamp per run) and six ice bricks (two stacked on top of one another beneath the metal base for each run) to cool the other end. Ice-bricks were small (165 x 950 x 20 mm) hard plastic water receptacles that could be frozen (typically used to keep food cold).

To achieve a suitable thermal environment, thermal gradients were tested and calibrated over a period of a week prior to the commencement of experiments. This involved recording temperature in the gradients by placing individual iButton® temperature data loggers (Maxim Integrated, San Jose, California) spaced 24 cm apart down the length of the run for a period of 24 hr. Tests were run in each of the different gradients to ensure no differences between them. To delay melting, the ice bricks were insulated on the bottom and sides with polystyrene, leaving the top open to be in direct contact with the metal base of the thermal gradients. By changing the ice bricks every 8 hr, the cool end of the gradient was kept stable. Tests with iButton® data loggers demonstrated a temperature gradient ranging from 17–62 °C was achieved. The lighting regime in the

laboratory followed a day/night light cycle that was very similar to what these nocturnal geckos would experience in the field (i.e. dark most of the time). The windows of the demountable building were blocked with curtains to reduce incident sunlight. At night, room lighting was limited to short periods only when required. An iButton® was also placed outside the gradient during tests to confirm room temperature remained with one degree of 20 °C.

Thermal preference experiments were typically commenced within 1 hr of returning the geckos to the laboratory after capture. Each gecko was placed in the centre of the thermal gradient and its entry time noted. Thermal preference experiments were conducted for 24 hr, during which two temperature measurements were taken: one at midday to represent day-time preference ($Tpref_{mid}$) and another after 24 hr for night-time preference ($Tpref_{24}$). These temperatures were classified as 'preferred' body temperatures ($Tpref$). $Tpref_{mid}$ was measured between 12.00 hr and 13.00 hr using the Fluke® infrared thermometer (distance to spot ratio of 30:1) pointed at the gecko's lower back and base of tail, close to the cloaca at a distance of approximately 5 cm to give an infrared spot size of approximately 2 mm. Infrared measurements were taken at this mid-point of the experiment (versus determining the cloacal temperature) to minimise disturbance to the geckos. $Tpref_{24}$ was measured by inserting the thermocouple of a Fluke® 566 infrared thermometer (Fluke Australia Pty Ltd, Castle Hill, NSW) in the cloaca after each gecko had been in the thermal gradient for 24 hr. Additionally, at the time of $Tpref_{24}$ measurement, the orientation of each gecko's tail was noted with respect to the heat source. Gecko tails classified as pointing 'toward' were closer to the heat source than the head. Accordingly, tails classified as 'away' were further than the head from the heat source.

Body measurements were taken at the conclusion of thermal preference experiments. Snout-vent length was measured to the nearest 1 mm using a clear plastic ruler. Body mass was measured to the nearest 0.1 gm using Pesola® spring scales (Pesola AG, Baar, Switzerland). Additionally, whether the tail was original or regrown was noted. Regrown tails are easily identified by the absence of the characteristic caudal knob (Russell & Bauer 1987). After measurements were completed, the geckos were again placed in individual cloth bags and stored in an ice-chilled cool box. Each gecko was then transported to, and released within, a few metres of the original flagged capture location.

Analysis

Comparison of gecko distributions between habitats and gecko tail orientations after 24 hr in the thermal gradients were conducted using a Chi-square Goodness of fit Test at $\alpha = 0.05$ in SigmaPlot Version 11 (Systat Software, Inc., San Jose, California). Relationships between field body temperatures and mass and field body temperatures and air and ground temperatures were assessed using linear regression in SigmaPlot Version 11 as well. Both temperature and mass data for the two species were combined for these analyses. All other statistical tests were conducted using the PROC GLM procedure

in SAS 9.4 (SAS Institute, Inc., Cary, North Carolina), with all comparisons made at $\alpha = 0.05$. Because of the imbalance in sample sizes (numbers of animals collected per species, by sampling date, or in different habitats), we used the Type III Sum of Squares in determining significance in the Analysis of Variance (ANOVA) tests used. Gecko morphometric parameters were compared using a one-factor ANOVA with species as the main effect followed by the Tukey method for post-hoc comparisons. Ground and air temperatures were similarly compared using a two-factor ANOVA with month and habitat as the main effects. Gecko field body temperatures between species were compared using an Analysis of Covariance with mass as the covariate, and differences between field body temperatures and preferred body temperatures at midday and 24 h ($Tpref_{mid}$ and $Tpref_{24}$) were determined using a repeated measures ANOVA.

Results

General habitat use and body size of the test species

A total of 27 *N. levis* and 52 *N. laevissimus* were captured within Uluru Kata-Tjuta National Park during the surveys reported here. The geckos demonstrated a significant difference in habitat use ($\chi^2 = 53.3$, $p < 0.001$), with *N. levis* primarily found in *Spinifex* sandplains (63%), mulga shrublands (30%), and occasionally on sand dunes (7%). In contrast, *N. laevissimus* was mostly found around sand dunes (90%), and occasionally in *Spinifex* sandplains (10%) in areas of open sand within short distances of a *Spinifex* tussock.

The average mass and SVL for *N. levis* (17.2 gm and 86.3 mm, respectively) were both significantly greater ($p<0.0001$) than those for *N. laevissimus* (7.3 gm and 67.6 mm, respectively, Table 1. A linear regression of field body temperature versus mass for all animals collected indicated a significant relationship ($p=0.019$, Body temperature = 22.8 + (0.198*Mass)), although the r^2 value of 0.08 indicates mass did not explain much of the variation in body temperature. The average field body temperature for *N. levis* measured for the animals over the course of the study (26.2 °C) was also higher than that for *N. laevissimus* (24.5 °C) but this difference was not significant based on an ANCOVA with mass as the covariate. It is interesting to note that while the maximum field body temperature measurements were similar between the species, *N. laevissimus* had a lower minimum recorded temperature which could account for the lower overall average body temperature for the species.

Table 1. Average morphometric and field body temperatures across all sampling dates for *Nephurus levis* and *N. laevissimus*. Numbers in parentheses are 1 standard error of the mean and the range of values recorded.

	Nephrurus levis	*Nephrurus laevissimus*
Mass (g)	17.2 (1.1, 9.0–25.6)	7.3 (0.5, 1.5–15.0)$^{*p<0.0001}$
SVL (mm)	86.3 (2.4, 62.0–112.0)	67.6 (1.7, 41.0–86.0)$^{*p<0.0001}$
Field body temp (°C)	26.0 (0.8, 19.5–33.5)	24.5 (0.6, 14.5–33.6)

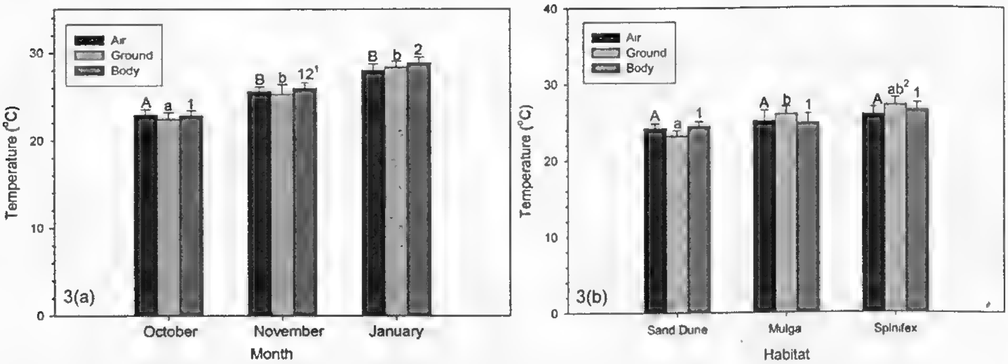

Figure 3. Average air, ground and gecko body temperatures based on sampling month (a) and habitat (b). For air temperature, bars that do not share a common upper-case letter are significantly different at $\alpha=0.05$, for ground temperature significant differences are indicated by bars that do not share a lower case letter, and for body temperature significant differences are indicated by bars that do not share a common number. Error bars represent 1 standard error of the mean. [1]p-value for November vs. January body temperatures = 0.0576, [2]p-value for Sand vs. *Spinifex* ground temperature = 0.0519.

We also compared air temperature, ground temperature, and combined field body temperatures within and across the sampling months of October, November, and January (Figure 3a), and within and between habitats (Figure 3b). There were no significant interactions between any combination of sampling month or habitat for any temperature variables. There was also no significant effect of species on field body temperature. Air and ground temperatures were significantly cooler in October as compared to November and January (Air temp: October vs November, $p=0.033$; October versus January, $p<0.0001$; Ground temp: October versus November, $p=0.027$; October versus January $p<0.0001$). Average field body temperature was significantly lower in October than in January (22.8 versus 28.8, $p<0.0001$). Field body temperature was also lower in November as compared to January, but this was just on the limit of statistical significance (25.9 versus 28.8, $p=0.0576$). There were no significant differences in air or body temperatures between the three habitat types, but ground temperature in the sand dune habitat was significantly lower than that in Mulga (23.2 versus 26.0, $p=0.0324$) and nearly significantly lower than that in *Spinifex* (23.2 versus 26.5, $p=0.0519$).

To further evaluate the degree of thermoconformity the geckos exhibited to environmental temperatures, the combined field body temperature data for both species were regressed against air (Figure 4a) and ground temperatures (Figure 4b). Significant positive relationships were detected between body temperature and both air ($r^2 = 0.932$, $p < 0.001$) and ground temperature ($r^2 = 0.820$, $p < 0.001$), with field body temperature displaying a slightly stronger relationship with air temperature.

Thermal preferences

For both *N. levis* and *N. laevissimus*, the average $T\textit{pref}_{\mathrm{mid}}$ of 29.1 °C (range: 24.6–38.7 °C), and 29.4 °C (range: 20.9–34.6 °C, respectively, was significantly greater than the average Tb (Figure 5). Similarly, the average $T\textit{pref}_{24}$ of 28.1 °C (range: 24.5–32.5 °C) and 27.8 °C

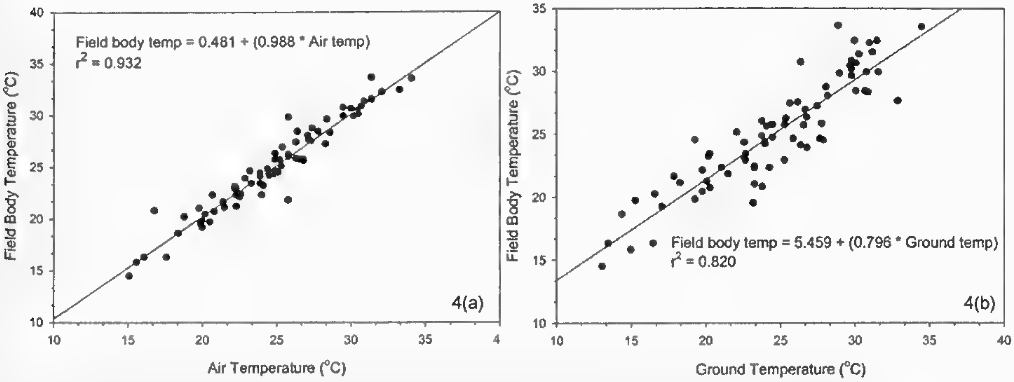

Figure 4. Relationship between field-active body temperature and air (a) or ground (b) temperature at place of capture across all habitats for both *Nephurus levis* and N. *laevissimus.*

(range: 20.9–34.6 °C), respectively, were also significantly greater than their T*b* (P<0.0001 for all comparisons except T*pref*$_{24}$ vs. field body temp for *N. levis,* for which p=0.0002). No significant differences were detected in the T*pref* measurements between each species.

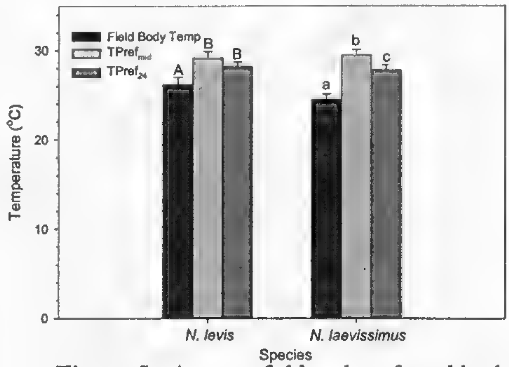

Figure 5. Average field and preferred body temperatures at midday and 24 h for *Nephurus levis* and *N. laevissimus.* For *N. levis,* bars that do not share a common upper-case letter are significantly different at α=0.05. For *N. laevissimus* significant differences are indicated by bars that do not share a lower-case letter. Error bars represent 1 standard error of the mean.

Tail orientation and autotomy

To examine the potential role of the tail in assessing the thermal environment, tail orientation at the time the T*pref* measurements were taken was recorded. Orientation of the tail was classified as either pointing 'away' or 'toward' the heat source, with the expected frequency based on an equal number of individuals between the two orientations. No significant difference was detected between observed and expected tail orientations for either *N. levis* or *N. laevissimus.* However, for both species, approximately 67% of individuals orientated their tails 'toward' the heat source (Table 2). Analysis of data for both species combined demonstrated a significant difference between the observed frequency of individuals orienting their tail toward the heat source and the expected frequency (x^2 (1) = 5.56, p $\leq$0.05*).

Evidence of autotomy (i.e. absence of caudal knob) was apparent in 28% of *N. levis* captured. The selection of T*pref* by *N. levis* with regrown tails was not significantly different to geckos with original tails. This comparison could not be conducted for *N. laevissimus* because no captured individuals of this species displayed evidence of tail autotomy.

Table 2. Observed tail orientation after 24 h in the thermal gradient for both *Nephurus levis* and *N. laevissimus* individually and combined. Orientation of tail was classified as either pointing 'away' or 'toward' the heat source. Critical values from Chi-Square Goodness of Fit test and associated P-values are based on expected equivalent distributions of individuals in the two orientation categories.

	Nephrurus levis		*Nephrurus laevissimus*		*Combined*	
	Toward	Away	Toward	Away	Toward	Away
Observed	13	6	18	9	31	15
Expected	8.5	8.5	13.5	13.5	23	23
Critical, p-value	3.11, 0.077		3.00, 0.083		5.57, 0.018*	

* Statistically significant at $\alpha = 0.05$.

Discussion

Field body temperatures and thermal preferences

Statistically significant relationships between field-active body temperature (T*b*) and environmental temperatures measured at the capture location (T*a* and T*g*) in this study suggest that these two *Nephrurus* species may not be behaviourally thermoregulating at night. The strong relationships meet Huey & Slatkin's (1976) criteria for thermoconformity and suggest that both *N. levis* and *N. laevissimus* may be mostly thermoconforming at the prevailing environmental temperatures for this study. This is consistent with previous studies describing thermoconformity during activity by *Nephrurus* congeners (Licht *et al.* 1966; Pianka & Pianka 1976) and other nocturnal ectotherms including other gecko species, lizards (e.g. *Zootoca*) and iguanas (e.g. *Sphenodon*) (Huey 1974; Herczeg *et al.* 2003; Wu *et al.* 2009; Tan & Schwanz 2015). One limitation to this conclusion, however, is that this study did not measure the range of available microhabitat temperatures at each capture location, which is necessary for a truly comprehensive description of temperature regulation (Hertz *et al.* 1993). Thermoregulation at night has previously been reported for nocturnal thigmotherms (Rock & Cree 2008; Tan & Schwanz 2015), although the high energetic cost of behavioural thermoregulation is suggested to be a factor influencing its use (Huey 1974).

T*pref* selected by *Nephrurus* geckos in the thermal gradients were similar to those reported for other gecko species (Angilletta and Werner 1998; Angilletta *et al.* 1999; Kearney & Predavec 2000; Hitchcock & McBrayer 2006; Tan & Schwanz 2015), although T*pref* for both species were higher than both the day (24.2 °C) and night (26 °C) T*pref* reported for *N. stellatus* (Angilletta & Werner 1998). One potential problem with our T*pref* estimates is that they are based on single measurements of the geckos taken at midday and midnight. As such, we may be presenting a body temperature of the animal in some phase of behavioural thermoregulation rather than a truly "preferred" temperature. Other studies that have investigated thermal preferences in geckos have relied on multiple measurements of body or substrate temperature taken over the course of the time they are in the thermal gradients (e.g. Angilletta & Werner 1998; Angilletta *et al.* 1999; Tan &

Schwanz 2015). While the movement patterns of geckos in this study were not closely monitored, the majority of individuals were often found partially buried in the thin layer of sand of the thermal gradient when the temperature measurements were taken, indicating they had been in place for at least some period of time. Additionally, the variability in the T*pref* temperatures reported here are comparable to that reported in the previously cited studies that used continuous monitoring (e.g. 1 SE< 1) and the temperature ranges are also comparable to those studies in which range was reported. We therefore feel our reported temperatures are reasonable approximations of a preferred body temperature for the geckos.

While *N. levis* and *N. aevissimus* did not significantly differ in T*pref*, *N. levis* displayed a significantly higher mean T*b* (24.7 ˚C) than *N. laevissimus* (22.3 ˚C). This difference may be driven by substrate differences between the habitat types. Although we found no difference in air temperature between habitats, the average ground temperature in sand dunes was significantly lower than in Mulga and nearly significantly lower than that for *Spinifex* sandplain habitats. Sand dunes have soft, sandy soil, which is known to be more thermally diffusive than the clayey sands and soils of sandplains and shrublands (De Silans *et al.* 1996), therefore leading to more rapid loss of heat. The lower minimum T*b* recorded for *N. laevissimus* may reflect the lower substrate temperatures in the sand dunes where this species was common and in turn the lower average T*b*. T*b* measurements for *N. levis* and *N. laevissimus* were lower and more variable than both $Tpref_{mid}$ and $Tpref_{24}$, which is consistent with the idea of the nocturnal paradox. These results are also consistent with previous studies comparing T*b* and T*pref* (Huey & Bennett 1987; Angilletta & Werner 1998; Gil *et al.* 1994; Angilletta *et al.* 1999; Hitchcock and McBrayer 2006; Wu *et al.* 2009; Tan & Schwanz 2015).

However, the paradox discussion is generally based around body temperatures that are optimal for locomotion, as that is considered to be one of the most important physiological functions during activity (Autumn *et al.* 2002). Our study measured the body temperatures selected by *Nephrurus* geckos in a thermal gradient and it is unknown how these may compare to body temperatures optimal for sprint performance. In diurnal lizards, these two temperatures have been reported as being very similar (Huey & Bennet 1987; Angilletta *et al.* 2002; Lailvaux *et al.* 2003). Although the nocturnal and diurnal lizards studied by Huey & Bennet (1987) exhibited a similar optimal sprint performance body temperature, their nocturnal subject species, including Royal Ctenotus (*Ctenotus regius*), Copper-tailed Skink (*C. taeniolatus*) and Spotted Ctenotus (*C. uber*), actively selected a much lower T*pref* in the thermal gradient. This suggests that the selection of T*pref* in lizards may not always prioritise locomotion.

Most nocturnal ectotherms thermoconform to environmental temperatures during activity due to limited thermal variation (Pianka & Pianka 1976; Licht *et al.* 1966; Herczeg *et al.* 2003; Hitchcock & McBrayer 2006; Wu *et al.* 2009; Tan and Schwanz 2015). Therefore, behavioural thermoregulation may be largely confined to daytime refuge sites as the soil offers a vertical thermal profile (Körtner *et al.* 2008). It is possible

that selection of body temperatures during refuge would more likely reflect priorities of metabolism, digestion and growth than sprint speed (Kearney & Predavec 2000). This notion is supported by Carretero *et al.* (2005) who suggest that temperature preference should be interpreted as a compromise between selective pressures, both from the thermal environment and energy allocations/requirements. To better understand the physiological significance of these T*pref* exhibited by *N. levis and N. laevissimus*, further work is required involving testing optimal temperatures for various physiological and behavioural functions and evaluating these based on measured temperatures in daytime burrows.

Tail orientation and autotomy

The only extensive investigation of the *Nephrurus* tail knob, including histological and anatomical characterisations, was conducted by Russell & Bauer (1987). They concluded it had a probable functional role in monitoring the thermal environment. However, to date there has been no other strong evidence to explain the function of the unique tail morphology. Analysis of tail orientation by *N. levis* and *N. laevissimus* geckos in the thermal gradient suggests that *Nephrurus* geckos are selectively positioning their caudal knob towards the heat source similar to the direction they are reported to be orientated within their burrow during the day (Pianka & Pianka 1976). These results may provide evidence to support the idea that the tail plays some role in assessing the thermal environment, although just what that role might be is unclear.

Autotomy frequencies in *Nephrurus* are low by gecko standards. Russell & Bauer (1987) and Pianka & Pianka (1976) found that *N. laevissimus* had the lowest (0.6%) reported frequency of the 12 nocturnal lizards they investigated in the Western Australian desert. The present study supports these observations, with none of the 52 *N. laevissimus* captured exhibiting signs of autotomy and only a limited number of *N. levis* exhibiting it. The frequency of tail autotomy is said to be driven by both the number of autotomy sites in the tail and the cost of tail loss to the individual (Bateman & Fleming 2009). *Nephrurus* exhibits only a basal autotomy site which may indicate the cost of losing the tail outweighs the benefit of its loss. This may support the idea that the tail knob plays some important physiological role. Regrown tails are reported to show skeletal, neural, and muscular differences as compared to original tails (Bellairs & Bryant 1985), which could influence sensory capacity of the structure if such a role exists. However, we found no significant difference in T*pref* between *N. levis* with original and regrown tails.

Summary

The results of this study indicate that *N. levis and N. laevissimus* exhibit size and habitat use differences that may lead to a slight difference in field body temperature. However, we demonstrate that both species are predominantly thermoconforming during activity and often do not reach their T*pref*, that is, they exhibit the 'nocturnal paradox'. This study adds recent and relevant data to a relatively dated body of literature on thermal preference in nocturnal reptiles. Further investigation of the thermal environment of

the daytime burrow for these species and the associated body temperatures therein provide an interesting area for further research. Individuals from both species also had a greater propensity to orient the tail knob toward the heat source in the thermal gradient experiments, although no temperature differences were observed between geckos that had lost and regrown tails and those that had not. The frequency of tail autotomy was low in field-collected animals and the role this structure may play in the thermal biology of these organisms remains unclear.

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An observation of Burton's Legless Lizard (*Lialis burtonis*) in a tropical mangrove forest

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Abstract

A Burton's Legless Lizard (*Lialis burtonis*) was observed within the *Rhizophora* zone of a tropical mangrove forest in the Top End of northern Australia in November 2018. To the author's knowledge, this is the first record of *L. burtonis* within a tidal forest. Although this lizard occurs in virtually all habitat types and is capable of comparatively long-distance movements, its presence in mangrove forests is presumably a rare occurrence.

Burton's Legless Lizard (*Lialis burtonis*) is a distinctive, snake-like, functionally limbless squamate of the family Pygopodidae. The species is a sedentary ambush predator (Patchell & Shine 1986; Murray *et al.* 1991) that feeds on lizards, primarily skinks (Wall & Shine 2013), but it also takes geckos, agamids, other pygopodids and small snakes (McKay 2017; Wilson & Swan 2017). *Lialis burtonis* is Australia's most widespread lizard, inhabiting virtually all terrestrial environments from desert interiors to the margins of wet rainforests (Cogger 2000; Wilson & Swan 2017). Based on the available literature and anecdotal evidence, there are no accounts of *L. burtonis* utilising mangroves. Herein I report, what is to my knowledge, the first observation of *L. burtonis* within a tropical mangrove forest.

During a vegetation ground-truthing survey on 21 November 2018, an adult *L. burtonis* (334 mm snout-tail length) was observed actively crawling on the mud surface in a closed, *Rhizophora stylosa*-dominated mangrove forest (Figures 1, 2). The site was adjacent to a river bank along a tidal section of the lower Blackmore River, Northern Territory (location: 12° 43.309'S, 130° 56.646'E). The lizard was observed at 13.54 hr and was located approximately 27 m from the river bank, 22 m seaward of the mean high-water neap tide line (at the interface between the *Rhizophora* and *Ceriops* mangrove zones), and 98 m from the nearest landward margin with terrestrial vegetation. The body colouration of the lizard consisted of a pale cream background colour, dark pigment along the face and fore-body, and a bold white lower lateral stripe running from the snout (Figure 3).

Despite being a sedentary ('sit-and-wait') ambush predator, *L. burtonis* periodically makes comparatively long-distance movements. In the Top End of the Northern Territory, Wall & Shine (2013) showed that although *L. burtonis* moves on average 4.9 m per day, but usually much less, some individuals can travel up to 40 m in a single foray. These longer moves are typically conducted at night and are likely instigated by the need to

Figures 1–3. Burton's Legless Lizard (*Lialis burtonis*) recorded from the Blackmore River mangrove site. **1.** Individual actively crawling on the mangrove forest floor **2.** Observation site comprised of a tall-closed *Rhizophora stylosa* mangrove forest. **3.** Individual displaying the distinctive patterning and colouration on the head and fore-body. (Adam Bourke)

.find a new foraging site (Wall & Shine 2013). Perhaps it was an errant, long-distance movement into the mangroves from the adjacent terrestrial habitat that explains the observation presented here.

Mangrove forests are undoubtedly marginal habitats for *L. burtonis* as tidal flooding is problematic, if not lethal. The climbing of mangrove roots and trunks would be testing for pygopodid lizards, as the body of *L. burtonis* is far less supple than those of snakes (M. Wall pers. comm.). Additionally, suitable prey items required by *L. burtonis* are scarce in mangrove forests; the only small lizards inhabiting local mangals are *Cryptoblepharus* skinks and an indeterminate species of gecko (pers. obs.). As such, mangrove forests are likely to be unsuitable habitat for *L. burtonis.* Thus, the presence of the species within mangrove forests is presumably a rare occurrence, one resulting from misadventure rather than an intentional and deliberate relocation.

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Sex determination in-ovo as a population management tool for the Southern Cassowary and Coastal Emu

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Abstract

Australia has lost two species and one subspecies of ratite due to historical persecution and anthropogenic changes to their habitats. Two additional ratites, the Southern Cassowary *(Casuarius casuarius johnsonii)* and Coastal Emu *(Dromaius novaehollandiae)*, are following this trajectory and will require human intervention to prevent further population decline or extinction. Advanced reproductive technologies offer promising avenues for the management of genetic and demographic diversity, both in-situ and ex-situ. Here, we review two important techniques and their efficacy in the conservation management of Australian ratites. The first is the sexing of an embryo in-ovo. This technique is central to developing technologies, however, is presently problematic in species producing thick and heavily pigmented eggs (i.e. Cassowary and Emu). Developments including the reduction of the shell membrane may promote light refraction and thus render the technique able to be used safely with developing embryos of Australian ratites. The supplementation of natural hormone during early embryonic development is another tool for sex regulation in ratite embryos. This was previously trialled in the Ostrich to influence a preferred gonadal sex, however, it will be trialled in the Emu and Cassowary for the first time. The use of these advanced reproductive technologies will complement current conservation efforts for Australian ratites, and facilitate the establishment and maintenance of robust captive populations.

Introduction

Since European settlement, Australia's human population has increased from an estimated 750,000 (ABS 2008) to over 25 million people (ABS 2018) and the forested areas have almost halved (DEE 2016). During this same period, Australia has suffered the extinction of approximately 60 vertebrate species (EPA 2018), with 17 additional species facing a greater chance than not of becoming extinct within the next two decades (Geyle *et al.* 2018). Australia faces a distinct confluence of threatening processes that are consistent across taxonomic groups, including the introduction of invasive species, habitat modification and agriculture (Kearney *et al.* 2018). The enduring nature of these threats across large landscapes will threaten the persistence of additional Australian

wildlife populations, including its ratites, causing more to become conservation-reliant. A balanced resolution is required in which both human and wildlife needs are met. Applied population management is one of the many tools required in the framework for such a resolution.

Populations that become isolated due to habitat fragmentation progressively require an integrated management approach to maintain genetic health and endure stochastic events. The periodic exchange of animals within and between natural (in-situ) and captive (ex-situ) environments, can improve the genetic diversity of both populations, and reduce the need for either of them to maintain such large numbers (Willis & Wiese 1993; Lacy 2013). Current Australian examples of this include the Helmeted Honeyeater (*Lichenostomus melanops cassidix*), Regent Honeyeater (*Anthochaera phrygia*) and Orange-bellied Parrot (*Neophema chrysogaster*). Although the Cassowary and Emu each present unique management challenges, such transactions would complement other vital components of their respective species management strategies.

In-situ management

To effectively manage a wild population, a species must be recognised not as a discrete entity but as part of a complex ecological system. Most in-situ management efforts including those for the Cassowary and Coastal Emu, therefore, direct greatest funding towards managing remaining habitat, often with the benefit of protecting numerous other species that occupy the same space. Despite these efforts, an increasing number of species are becoming conservation-reliant (Scott *et al.* 2010; Goble *et al.* 2012), requiring species-specific management via direct and ongoing human intervention (Hartl *et al.* 2014).

The success of in-situ programs require an intricate knowledge of a target species. Understanding the behaviour, demographics, movement ecology, social requirements, diet, reproduction and development and other physiological factors is critical for identifying threatening processes, and thus their successful management. The captive environment may therefore serve not only by supplementing wild populations, but also provide a space in which to gather some of this information.

Ex-situ management

Managing an in-situ population cannot be done without considering the community or ecosystem as a whole. However, managing an ex-situ population offers an opportunity to isolate species-specific factors that must be studied or considered, without the high level of unpredictability or complexity of surrounding natural environments. The captive environment can thus provide a safe platform to develop and test new ideas for conserving species. It allows the generation of hypotheses in a comparatively constant environment where certain predictions can be tested and repeated in a formal process. Concern often surrounds the establishment of captive populations due to the risk of

draining already meagre government funding away from in-situ priorities. Most ex-situ initiatives, however, operate on minimal government funding with many participating facilities contributing solely in-kind or through grants offered by the private sector.

While there are key benefits to ex-situ management, there are also concerns. An artefact of long-term and isolated ex-situ populations is the adaptation of a species to captivity (Frankham *et al.* 2010). Similarly, individuals set for wild release after captive breeding can also be naïve to the dangers of natural environments. Enclosure availability, inter- or intraspecific compatibility, sex skews and access to new founders all present limitations in a captive setting, however none are unique in theory to their wild counterparts. Recent scientific and evidence-based techniques have helped immensely with dealing with these challenges of captivity. Such improvements include population management software, the development of comprehensive husbandry manuals, providing appropriately stimulating environments, and training or otherwise preparing captive individuals for release (Mäekivi 2018). One simple example is 'ghosting', where a sheet is worn whilst hand feeding chicks to reduce humanisation (Gage & Duerr 2007). A more complex example is that of monitoring the reproductive status and ovulation of a bird (Jensen & Durrant 2006) for artificial insemination. Such advancements enable the ex-situ management of species to be conducted at the high standard required for the maintenance of insurance populations separate from those in the wild – a valuable component of many integrated management strategies.

It is therefore clear that every threatened species presents unique challenges for successful management. In this paper, we present and review two techniques that may be used in the intensive management of avian species that produce eggs with thick or heavily pigmented shells – sexing in-ovo and hormonal supplementation during early embryonic development. In Australia, this involves two endemic ratites, the endangered Southern Cassowary (*Casuarius casuarius johnsonii*) and an endangered population of the Emu (*Dromaius novaehollandiae*), herein referred to as the Coastal Emu. We will first introduce the issue of population sex ratio skews in general, then discuss management techniques for the egg stages of these birds that focus on sex ratio management and reproductive interventions.

Population sex ratios

Sex ratios are an important consideration for the stability of both in- and ex-situ populations. While an offspring ratio of 50:50 male:female is often assumed to be most natural or stable, this is not always the case. Different species exhibit widely variable stable sex ratios, which can be observed at hatching or can occur at a later life stage because of sex-specific mortality. In a review of sex ratios in 200 adult wild bird populations, 65% differed significantly from parity, with distortion 'significantly more severe' in threatened species (Donald 2007). In small or isolated conservation-reliant populations, the ability to manage sex ratios prior to hatching may become a crucial component for the management of genetic and social diversity and health.

Sex skews as threatening processes

When an unfavourable sex skew (especially towards many males) is considerable, there are obvious consequences for population growth. While males may fertilise and therefore potentially father hundreds of offspring, their success is limited by the number of receptive females. However, when parental care roles are reversed, such as with the Cassowary and Emu, so too does this paradigm. Therefore, although a sex skew towards females does not necessarily limit most bird populations, in paternal care species, such as the ratites, it may.

There are countless environmental and physiological variables that may lead to an unfavourable skewed sex ratio in-situ. A significant and adverse skew exists in one of Australia's most endangered birds, the Orange-bellied Parrot (*Neophema chrysogaster*). The future of this species is in peril, in part, due to the highly skewed in- and ex-situ populations (Stojanovic *et al.* 2018). The 2018 count of wild birds recorded 16 males and only 2 females (Dalton 2018), forcing the population to be dependent on the release of captive-bred birds for its survival (Heathcote 2018; Stojanovic *et al.* 2018). Skewed ex-situ populations of Columbiformes are also common, such as the White-throated Ground-dove (*Gallicolumba xanthonura*), which at one institution rarely produced females (Hall 2011), and a female Eclectus Parrot (*Eclectus roratus*) which produced 20 consecutive male offspring (Heinsohn 2008). Sex skews in captive populations have been attributed to high quality diets where females in good condition produce the most physiologically expensive or resource-dependent sex (Bradbury & Blakey 1998; Henderson *et al.* 2014). Similarly, the supplementary feeding of wild Kakapo (*Strigops habroptilus*) caused a significant excess of male offspring (which weigh 30–40% more) compared to that of non-supplemented females (Clout *et al.* 2002). Elevated levels of the stress hormone corticosterone have also been commonly identified as a potential mechanism for sex ratio adjustment, causing females to produce significantly female-biased broods in quails (Pike & Petrie 2006), sparrows (Bonier *et al.* 2007), finches (Gam *et al.* 2011), peafowls (Pike & Petrie 2005), and more males in chickens (Pinson *et al.* 2011). Stress hormones are particularly sensitive to environmental perturbation (Pike & Petrie 2006), and thus may serve as front line regulators for appropriate sex allocation. Sex skews can be identified and managed in captive populations, however its occurrence is concerning for threatened populations in-situ.

Sex skew management in birds

As shown by recent research, the mechanisms for determining the sex of an offspring across the animal kingdom are much more flexible than previously thought. This plasticity of sex is most commonly associated with reptiles, amphibians and fishes (Uguz *et al.* 2003), but has also been recorded in numerous bird species. In birds, it is the female that is the heterogametic sex, carrying both a Z and W chromosome (males carry ZZ) so any influence over sex may be under maternal control. Although the genetic sex of birds is determined at fertilisation, it is understood that environmental and physiological factors of the female may influence the final translation of sexual differentiation

throughout the body (Arnold & Itoh 2011; Ayers *et al.* 2013; Beukeboom & Perrin 2014; Major & Smith 2016). Conditions may prompt an investment in the preferential release of specifically-sexed oocytes (Rutkowska & Badyaev 2008) or may also occur post-fertilisation, requiring a sex reversal and creation of chimeric offspring (Major & Smith 2016). For example, a genetically determined male bird can otherwise appear to be female with functional ovaries, or a genetic female can appear as a phenotypic male with functional testes. Known as sex reversal, this phenomenon has been observed in the wild, and induced experimentally, in both embryos and in sexually mature individuals.

A physiological recognition of natural stressors such as food availability, followed by sex allocation, could impart strong fitness consequences (Tagirov & Rutkowska 2013). For example, in sexually dimorphic species, male offspring are often larger and grow at a faster rate compared to their female siblings (Gill *et al.* 1995). Male young would therefore require additional resources and be more likely to starve or be less competitive reproductively than female offspring if conditions were poor. Elevated levels of corticosterone is a common factor influencing sex in many orders of birds. Elevated corticosterone can be a chronic symptom of individual stress, thus populations under stress are at risk of developing sex skews. As environments change, sensitive species may find themselves under stress and therefore produce more of one sex.

Oviparous species such as birds present a unique opportunity to easily access embryos at an early stage of development, thus enabling management techniques to be applied much earlier than with other groups. Basic reproductive interventions are common practice in good zoos and are already part of established management programs. In the case of sex-skews and possible detrimental population sex ratios for example, the introduction of natural hormone during early embryonic development may mimic maternal expression and exploit this plasticity in bird sexual differentiation. This ability would result in unprecedented control over population sex ratios and promote optimal sustainability in threatened species.

In-ovo sexing

The ability to determine the sex of a developing avian embryo (in-ovo) first appeared in the literature in 1997 as an ex-situ technique (Langenberg *et al.* 1997) and it also represents a significant advancement for the management of threatened species in-situ. If the sex of an embryo can be identified early, the preferred sex may be fostered, and incubation of the undesired sex discontinued at an ethical stage of development. Applications can amend sex skews, assist in the tailored construction or reconstruction of a population, or facilitate the demand of a specific sex for housing, breeding or to address intraspecific challenges. Furthermore, sexing an embryo in-ovo is a useful tool in addressing stochastic or random processes of populations and demographic uncertainty (Dutton & Tieber 2001).

Sex identification in-ovo involves drawing blood from the egg, that can then be genetically tested (Langenberg *et al.* 1997; Dutton & Tieber 2001; Jensen *et al.* 2012). Identifying

vascular development in an egg is the precursor for extracting a blood sample for sexing. This is a straightforward method for most eggs, involving the use of a common candling apparatus to illuminate the vascular system (Figure 1a). Once identified, a small window is made in the shell to safely and accurately draw blood (Figure 1b). However, thick or heavily pigmented eggs prevent the penetration of light at a level that is necessary to highlight vascular development and safe for the developing embryo. This means that when using this method, sex determination of embryos inside heavily pigmented eggs is currently not possible.

We propose a new technique for thick or heavily pigmented eggs whereby the shell thickness is reduced in two places to allow for the refraction of light and for clear observation of the vitelline vessel (Figure 1c). An appropriately thick or truncated vessel can then be identified and marked on the shell membrane for follow-up blood draw. The viability and efficacy of this technique is currently being examined.

Hormone supplementation

During embryonic development, the sequence of gonadal differentiation may be changed by hormones or non-hormone inductor substances within the embryo (Baker 1972). Although the genetic sex remains constant, the female hormone estradiol-17β has been used by researchers to successfully induce a female phenotype from genetically male embryos (Major & Smith 2016) and an aromatase blocker (i.e. an enzyme that stops the transformation of testosterone to estradiol) has been used to induce a male phenotype from genetically female embryos (Vaillant *et al.* 2001; Takagi *et al.* 2007). Thus, we know that sex reversal by hormonal manipulation is possible, with no adverse health effects on the individual (Crews & Wibbles 1995; Dutton & Tieber 2001).

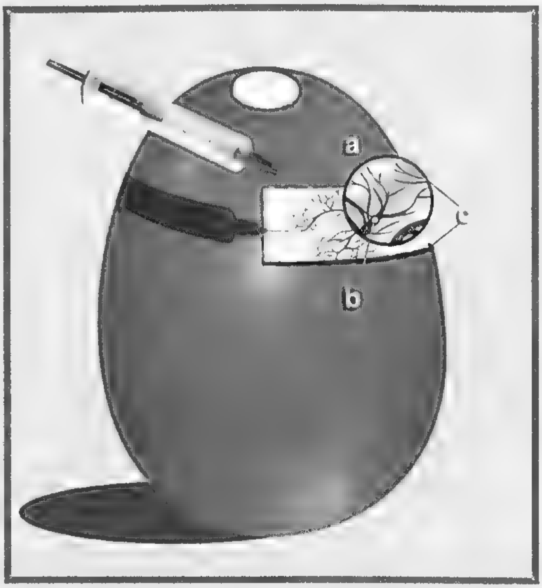

Figure 1. Details of proposed egg sexing procedure for thick or heavily pigmented eggs. **a,** highlighted vascular development, **b,** small window for blood draw, **c,** areas of shell reduction.

Sex reversal has previously been achieved as a method for the preferential production of Ostriches (*Struthio camelus*) (Crews & Wibbles 1995) with treated birds producing viable offspring when paired with normal birds (Crews & Wibbles 1995). These results are promising for the use of hormone-induced sex reversal in the management of Australia's ratite counterparts.

Although sex reversals have been observed in the wild and been achieved in ex-situ trials, there is much to consider before adopting the method in-situ, or in threatened species management. A major concern for conservation is the occurrence of birds that may, for example, be a physiological representation of a male but are behaviourally redundant in performing courtship displays or associated breeding behaviour. Conflicting sex hormone production in the brain and gonads may be responsible for such redundancies in male or female sexual behaviour.

Further research is required in the use of hormone supplementation to achieve sex reversal and follow-up monitoring at maturation of the breeding success and behaviour of resulting individuals. We will be trialling sex reversal in two species of ratite, using estradiol-17β to stimulate a female outcome and an aromatase blocker to achieve a male outcome. If successful in creating sex reversed birds that are behaviourally and sexually functional, these techniques may significantly improve ongoing conservation efforts of threatened ratite populations.

Xenotransfer

An emerging reproductive technology is the transfer of gonadal germ cells (GGCs) from threatened species into a common, sister species host. In a process named xenotransfer, GGCs are inserted into host eggs prior to sexual differentiation (Imus *et al.* 2014). The subsequent successful migration and colonisation of these cells in the gonadal ridge (i. e. the location of gonadal development) of the host occur within hours of injection (Roe *et al.* 2013; Imus *et al.* 2014). The resulting individuals can produce donor-derived sperm (Imus *et al.* 2014), that can be collected for future artificial insemination. Xenotransfer has also been successful at restoring the ability to produce sperm (spermatogenesis) in sterile adult birds (Trefil *et al.* 2010) and in rescuing adult germ stem cells of genetically valuable deceased birds (Roe *et al.* 2013).

Despite the early nature of this work, it poses a promising avenue for the preservation of genetic diversity in small or critical populations. In combination with in-ovo sexing and hormone supplementation, these technologies will have a significant impact on the population management of species both in- and ex-situ.

Case Study: Australian Ratites

Ratites form one of the most basal lineages of birds, with living representations (Ostrich, Rhea, Kiwi, Emu and Cassowary) distributed throughout the southern hemisphere. These ancient birds separated from a common Gondwanan ancestor approximately 65 million years ago and have therefore played an active role in the formation of the current Australian biological landscape. Although the two species occupy different habitats, the service they provide for ecosystem dynamics is similarly important (Stocker & Irvine 1983; McGrath & Bass 2001; Sales 2009). Thus, the disappearance of either the Emu or the Cassowary from the landscape may not have immediately observable impacts, however in time, effects would likely be significant and long-lasting.

In the science of wildlife conservation management there are certain characteristics of species that may predict extinction, and the Cassowary and Coastal Emu fulfil the top three of these characteristics (i.e. large body size, small populations and variable growth rates). Both birds are listed as endangered – the Cassowary under the *Environmental Protection and Biodiversity Conservation Act (1999)* and the Coastal Emu under the New South Wales *Threatened Species Conservation Act (1995)*. Coastal Emus are geographically isolated from their more common inland counterparts, with preliminary DNA analysis indicating they are genetically disjunct (NCC 2014). Therefore, an integrated and adaptive management approach involving both in-situ and ex-situ techniques will be required for their long-term survival.

Current in-situ species management plans focus on the in-situ protection and restoration of habitat for both the Southern Cassowary (Latch 2007) and the Coastal Emu (OEH 2018). However, with a steadily growing human population and inadequate legislation to prevent the piecemeal conversion of suitable habitat, the threatening processes for these species are not likely to dissipate. Without the additional assistance of reproductive technologies and establishment of well managed ex-situ insurance populations, managers may struggle to preserve adequate levels of genetic and demographic diversity.

Ex-situ populations of Southern Cassowaries are managed in Australian, North American and European zoos but no Coastal Emus (only Inland Emus) are known to exist in captivity. With the exception of artificial incubation and hand rearing, no in-situ or ex-situ reproductive technologies for the purposes of active population management are known for either species.

The following case studies present current and emerging reproductive technologies available for the conservation and population management of the Southern Cassowary and the Coastal Emu. The technologies discussed are not intended to replace in-situ management efforts but compliment them in creating sustainable populations. The strength of this research is in assisting ex-situ populations to serve as a genetic reservoir for wild birds in the event of continued population decline or catastrophic loss.

Southern Cassowary

The Southern Cassowary (Figure 2) is listed as Endangered under the Federal Government's *Environment Protection and Biodiversity Conservation Act* 1999 and managed over two separate regions of northern Queensland. For the purposes of this paper, the authors will concentrate on the southernmost Wet Tropics population that exists between Cooktown and (just north of) Townsville. The Wet Tropics population provides a sound case study for the Cassowary as existing threats are compounded by anthropogenic activities in the region and it is generally more accessible for the implementation and monitoring of assisted reproductive technologies. The employment of a conservation detector dog may also assist in locating injured or deceased threatened species, including the Cassowary and the endangered Mahogany Glider (*Petaurus gracillis*), in less accessible forested areas.

Figure 2. Southern Cassowary male 'Rocky' with a clutch of eggs at Australia Zoo. (James R. Biggs)

Less than 20% of critical coastal lowland habitat remains in the Wet Tropics region, much of which is fragmented. This has resulted in the isolation of at least 10 separate Cassowary populations within the Wet Tropics region, six of which are considered to have extreme current and potential threats (Latch 2007). Current conservation efforts for the most vulnerable populations could be stripped in the event of one or more severe tropical cyclones. Recommendations from post-cyclone research in the Wet Tropics region have called for disaster management planning to treat the protection and rehabilitation of threatened species in a similar vein to that of human livelihoods (Turton 2012).

Although some endemic mammal species have proved remarkably resilient to the impacts of cyclones, Cassowaries have been found to be highly vulnerable, especially in fragmented areas such as Mission Beach (Turton 2012). This was highlighted in the aftermath of Severe Tropical Cyclone Larry in 2006, wherein over 30% of Cassowaries from one region affected by the cyclone perished (Turton 2012); all dependent chicks were presumed killed and almost 20% of the surviving adults and sub-adults perished from starvation and disease in the 12 months following the cyclone (Moore 2007; Goosem *et al.* 2011). It was estimated to take 30–40 years to restore the forest canopy which was still repairing from Cyclone Winifred 20 years earlier (WTMA 2007) when the region was hit again in 2011 by Severe Tropical Cyclone Yasi. It is well reported that the intensity of severe tropical cyclones is predicted to increase with the effects of climate change (Garnett & Franklin 2014). Thus, it is realistic to predict the extirpation of entire Cassowary populations in future cyclone events. In light of this information, the debate that surrounds the involvement of ex-situ contributions should become less contentious.

An ex-situ population management program is in place for the Southern Cassowary and a detailed husbandry manual exists (Biggs 2013). Participating zoos are accredited by the regional Zoo and Aquarium Association of Australasia and endorse their annual report

and recommendations for breeding and animal transfers. Despite these administrative achievements, unpredictable reproductive success and non-recommended breeding continues to threaten future management options (Biggs 2018).

There are multiple scenarios in which the reproductive technologies described above might benefit Cassowary management. The following are three such examples:

Scenario 1 – Improve and maintain high levels of genetic diversity in ex-situ population

Australia's captive population of Southern Cassowary consists of a stable population of 65 birds (32 males and 33 females) with a genetic diversity of 89.3% (Biggs 2018), close to the 90% recommended to maintain genetic diversity over 100 years (Frankham *et al.* 2010). With current limitations in carrying capacity and successful breeding, the ex-situ population is reliant on one in-situ acquisition every five years to maintain an acceptable level of genetic diversity (Biggs 2018). Managers of in-situ and ex-situ populations alike, have an invested interest in facilitating this transfer.

Acquisition of wild birds into captivity has consisted of unreleasable birds from rehabilitation facilities. This should continue to be the case, however if the opportunity arose for the acquisition of eggs, this would also be beneficial. Chicks can be hand reared from eggs but subsequent young often respond poorly to breeding behaviours upon maturation (Romer 1997; Biggs 2013). Ideally, an egg would be placed with an incubating or broody male (Figure 2). Ratite males have been observed brooding inanimate objects such as rocks or even bricks. Although not yet trialled in ratites, injections of prolactin have been used to induce broodiness in Rock Pigeons (*Columba livia*) (Lahr & Riddle 1938), Barbary Doves (*Streptopelia risoria*) (Lehrman & Brody 1964) and Chickens (*Gallus gallus domesticus*) (Riddle *et al.* 1935), indicating that it may also be effective in developing this behaviour in Cassowaries.

It is most important to manage the sex ratio of an ex-situ population as the birds cannot naturally disperse and mortalities do not appear to be sex-specific. Cassowaries are long-lived and expensive to maintain ex-situ. Thus, it is imperative that sex ratios in captive populations are tailored to maximise carrying capacity and reproductive effort. The sexing of eggs in-ovo would enable managers to achieve this. The future use of gonadal xenotransfer also offers an excellent avenue for introducing founder genetics without the ethical deliberation of accessioning wild birds.

Scenario 2 – Permanent isolation of an in-situ population

Since the 1600s, more than 90% of all recorded extinctions have been from isolated populations (Newton 1998; Frankham *et al.* 2010). Small, isolated populations require regular monitoring for genetic health (Frankham *et al.* 2010), which was recently done for the Cassowary by DNA faecal analysis and camera-trap detection using visual lures (McLean *et al.* 2017). The use of computer projections (population viability analysis - PVA) might assist in identifying approptiate timing for genetic supplementation and the creation of defensible thresholds. These can then be used to decide when and where

to introduce new birds. If ex-situ birds are selected, they should be as close to the founder level as possible, with low inbreeding and high genetic diversity to ensure the highest level of fitness and ability to adapt to the new environment (Frankham *et al.* 2010). New birds, of either captive or wild origin, will require varying degrees of soft release to a new environment. In-ovo sexing may be used to introduce a specific sex via nest manipulations, or to address sex linked genetic problems. The future use of xenotransfer in the Cassowary is a promising option for isolated populations. Gonadal material from deceased individuals would preferrably be introduced to a new host in-ovo, or theoretically, to sterile adult individuals.

Scenario 3 – A Severe Tropical Cyclone decimates lowland coastal habitat and multiple cassowary populations

Securing gonadal tissue from deceased birds is highly desirable after a cyclonic event as this tissue may be cryopreserved for use in future reproductive technologies. Following Cyclone Larry, 82% of Cassowary deaths in the Mission Beach area occurred on roads as a result of vehicle strike as birds wandered in search of food (Moore & Moore unpublished data (in Turton 2012)). The collection of samples in a timely manner is therefore not only an important acitvity but also a realistic acheivement in some areas.

The xenotransfer of this tissue into future embryos would enable managers to restore populations that were genetically predisposed to the region. As the reconstruction or regeneration of forested areas progress, birds may be incrementally returned to the area with the option of supplementary feeding stations, similar to those used to support starving birds following Cyclone Yasi (Turton 2012). Selected birds would prefferably be rehabilitated individuals from the region, or occur via natural immigration. If this were not possible however, birds of a selected age class, sex and demographic may be solicited from either nearby wild populations, or captive populations, or both wild and captive populations.

It is preferable that new birds are introduced as eggs to benefit from the knowledge and rearing of a paternal male, however, the chance of synchronising hatching events is low. It may become a genetically viable option to swap an entire clutch with a neighboring population or from the ex-situ population which would also provide some leeway in timing for hatching chicks. If this were to occur, activity trackers might be placed on incubating males or a movement sensor placed within a fake egg to monitor their movements and incubation behaviour, as has been used in the conservation management of New Zealand's endangered Kakapo (*Nestor notabilis*) (DOC 2019). Trackers were used in wild Cassowary by Campbell *et al.* (2012) to acquire home range data. The safe attachment and follow up removal of these GPS telemetry devices provides proof of concept for this proposal. Cassowaries are territorial, with females only tolerating an overlap with male birds (Moore 2007). In-ovo sexing might be used to tailor the addition of a specific sex, or sex ratio to maximise the carrying capacity. As with Scenario 2 above, the individuals selected must have low inbreeding and high genetic diversity. Although

hormone supplementation is an option for sex specific management, more research is required before it is implemented in-situ.

Coastal Emu

Emus were once distributed throughout most of mainland Australia and Tasmania, however they are now largely absent from southeastern coastal and subcoastal regions and other densely settled areas (OEH 2018). Two Emu species and one subspecies are now listed as extinct in Australia. The first to disappear was the once-abundant King Island Emu (*Dromaius ater*) which was swiftly driven to extinction by seal hunters and their dogs at the turn of the 19th century, disappearing in just a few years by 1805 (Brasil 1914). Next was the Kangaroo Island Emu (*Dromaius baudinianus*), which is believed to have become extinct in 1927 after being reported in great numbers only eight years previously (Parker 1984) [A painting of this bird appeared in the previous issue of *Northern Territory Naturalist* (Hermes 2018).] The Tasmanian Emu (*Dromaius novaehollandiae diemenensis*) was the last island population to disappear. Although never recorded as 'abundant', it was last recorded in the wild in 1865, believed to have become extinct as a result of hunting and poaching of eggs (Dove 1926). Following this same trajectory is the endangered population of the Coastal Emu from the New South Wales North Coast Bioregion and Port Stephens local government area.

The Coastal Emu is recognised as an endangered population by the state government of New South Wales with recent estimates of only 40–100 birds (OEH 2018) (Figure 3). It is highly susceptible to stochastic threats including fragmentation of habitat, anthropogenic disturbance, inappropriate fire regimes and predation of young and eggs.

Figure 3. Coastal Emu male with chicks, northern coast New South Wales. (Penelope Smith)

The cumulative effects of which have led to a feedback loop of population decline known as an 'extinction vortex'. The establishment of a captive population is considered by the New South Wales Saving our Species (SOS) program to compliment the imperatives of in-situ conservation. The IUCN recommends a five-step decision-making process when considering ex-situ management to evaluate its appropriateness and document process clarity and transparency (McGowan *et al.* 2017). Although it is considered outside the scope of this research, it is an essential first step.

There is a danger of leaving the establishment of an ex-situ population too late (IUCN/SSC & CBSG 1987; Mills 2013), and this may be the case with the Coastal Emu. The recommended number of founding individuals for an ex-situ population is 20–30 to establish an effective genetic base (Frankham *et al.* 2010). If wild collection is required, this should occur when the wild population is still relatively large (McGowan *et al.* 2017). This allows the collection of birds with low inbreeding, and has less impact on wild populations (Frankham *et al.* 2010). The establishment of an ex-situ population is recommended when wild populations drop below 1000 individuals (IUCN/SSC & CBSG 1987). Although that time has long passed for the Coastal Emu, it is expected that any negative impacts of small collection size from the wild population will be outweighed by the positive impact on the overall conservation effort.

The following scenarios provide examples of how reproductive technologies may assist the conservation of the Coastal Emu through improved population management both in- and ex-situ.

Scenario 1 – Establishment of a captive population

An aim to acquire at least 10 wild founders for an ex-situ population is necessary to maintain the minimum acceptable levels of genetic diversity (Frankham *et al.* 2010). Although this is not likely to capture a broad spectrum of alleles present in the population, considering the small existing population and length of presumed isolation, the number may be sufficient.

Initially, there would be little discrimination as to the sex of acquired birds with efforts concentrated on obtaining the most genetically diverse sample. Acquisition of eggs is preferable due to the ease of transport and limited stress on the individual. Although the chance of finding an incubating male is slim, it may be improved with the employment of trained conservation detector dogs. Eggs removed during the laying sequence or early in the breeding season should encourage egg replacement or reclutching by femlaes. Upon the acquisition of wild eggs, a captive male Inland Emu may be stimulated to incubate an artificial clutch by the incremental placement of artificial eggs, with or without the use of prolactin injection. Meanwhile, wild eggs might simultaniously be artificially incubated for safety and to monitor weight loss. Artificial eggs would then be replaced with their viable counterparts during the early stages of hatching for follow up rearing. Artificial 'smart eggs' (DOC 2019) are a recent development used to mimic the movement and sounds of a developing chick. These would alert the incubating male

to an iminent hatching event, prior to the egg switch. In the absence of the paternal services of a male Emu, eggs may be artificially incubated and hand reared by zoo professionals, preferrably in the company of farmed Inland Emu foster clutch mates.

Following the maturation of founder birds, the program enters a growth phase where reproductive technologies could play a major role in creating a demographically secure population. Efforts to collect additional founders from unrelated or disjunct populations should continue to be a high priority during this time. It is known from farming Inland Emus that females will breed as long as they are in sight of a male. Thus, using in-ovo sexing techniques or after further research, hormone supplementation, small breeding units may be created of multiple females and one male. Embryonic gonadal tissue of undesired sexes could be cryopreserved for future use in xenotransfer and to further reduce changes to the founder gene pool during captivity. The use of artificial insemination technologies (Malecki *et al.* 2008) could be employed to ensure maximum genetic diversity amongst the ex-situ population with minimum cost and need to facilitate animal transfers. Presently, this would only be possible in an ex-situ environment as birds require some familiarisation with the technique.

Scenario 2 – Re-establishment or supplementation of a wild population

The end goal of ex-situ conservation of species is reintroduction in the hope of establishing a self-sustaining population or supplementing the genetic or demographic health of an existing population. There are, however, concerns for the genetic adaptation to captivity. Limiting the number of generations in captivity is essential in reducing these effects and may be lessened further by the reintroduction of founder genetics via xenotransfer in released birds. Natural selection will assist to reverse the effects of deleterious adaptations accrued in captivity (Frankham *et al.* 2010) as long as adequate fitness is present to persist through the process. The monitoring and genetic management of reintroduced populations would be ongoing until it numbered the 500 individuals recommended to be free from human support (Becker 1984). This would involve techniques as described for the Cassowary in Scenario 2 above.

Conclusion

Conservation science is complex, and as we consider the endless practical questions involved with each species or population, it may raise more questions than it is able to answer. Although species habitat continues to reduce in size, quality and connectivity, the techniques for managing them are improving. The success of introducing such techniques as part of the Southern Cassowary and Coastal Emu management programs will hinge upon the financial support, facilities and experience of individual accredited zoo facilities in Australia, and their governing body. It will also require improvements in communication between government authorities and zoo professionals and increases in federal funding for threatened species management. With the assistance of new and emerging reproductive technologies, the integrated management of robust in-situ and ex-situ populations will create a brighter future for Australian ratites.

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Do freshwater mussels exist on the Tiwi Islands?

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Abstract

Exhaustive searches through collections and literature indicates that apparently no member of the freshwater mussel family Hyriidae exists on the Tiwi Islands, off the central Northern Territory coastline. Nor apparently does any hyriid exist on Cobourg Peninsula on the adjacent mainland of western Arnhem Land. These absences are despite seemingly suitable habitats being present on these large nearshore islands and landmass at the present time. Other hyriids do exist on nearshore islands in northern Australia and elsewhere in Australasia. The reasons for these absences are discussed briefly, with the most likely being a more arid climate during the Last Glacial Maximum drying out the streams and wetlands in the region.

Introduction

The only family of freshwater mussels in Australia, Hyriidae, consists of six genera (*Alathryia, Cucumerunio, Hyridella, Lortiella, Velesunio* and *Westralunio*) with approximately 20 species (McMichael & Hiscock 1958; McMichael 1967; Smith 1992; Walker *et al.* 2014). Conventional literature posits that hyriids "are found in all parts of mainland Australia" (McMichael & Hiscock 1958: 487; Walker 1981a; Smith 1998; Walker 1998), as well as (northern) Tasmania, New Guinea, the Solomon Islands, New Zealand and South America (McMichael & Hiscock 1958; Boss 1982; Walker 1981a; Smith 1992, 1998; Bogan & Roe 2008; Graf *et al.* 2015). Clearly such generalisations do not take the permanent inland deserts into account, but hyriids exist wherever suitable river systems – or even semi-permanent water holes – are present. In general, the distribution of individual species is determined, more or less, by the geography of the river systems (Iredale 1943).

Only three genera of the Hyriidae are present in the northern Australian region (termed the Leichhardtian Fluvifaunal Area by McMichael & Hiscock (1958) and McMichael & Iredale (1959)), which extends westward from Queensland, across the north of the continent, to the Kimberley region of Western Australia – *Velesunio* (which is more widespread and more abundant in the habitats in which it exists), *Alathyria* and *Lortiella*.

In July and August 2018, the junior author visited the Museum and Art Gallery of the Northern Territory (MAGNT) to study the collection of hyriids (Figure 1), both

the species present in the mollusc collection and their distributions. During the course of this research both authors were surprised to notice that there were no records of hyriids from the Tiwi Islands. Following this visit we sought information from other sources (mostly museums and databases) in Australia and overseas to decide if this were a genuine absence or merely a lack of sampling. In doing so, we also enquired about records for the Cobourg Peninsula, which is the closest major geographical feature on the Australian mainland coast to the Tiwi Islands.

Enquiries and Results

We undertook the following investigations to decide if any freshwater mussels existed at present, or historically, on the Tiwi Islands or Cobourg Peninsula:

- On 9 August 2018, RCW approached Vince Kessner, the person with the greatest knowledge on the non-marine molluscs of the Northern Territory. Mr Kessner had managed the health clinic at Milikapiti, on Melville Island, for some years and knows the Tiwi fauna very well. On the same day, Mr Kessner replied that no hyriids were present.

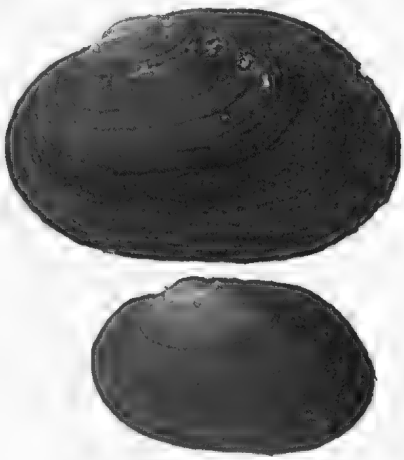

Figure 1. Specimens of *Velesunio* cf. *angasi* from Oenpelli Billabong (western Arnhem Land), which is apparently close to the point at which all freshwater mussels cease to exist on the central coast of the Northern Territory. Museum and Art Gallery of the Northern Territory, P.1076. Shell lengths 79.4, 57.3 mm. (MAGNT)

- On 9 August 2018, through Kate Hadden, RCW approached the 10 senior landholders from the Tiwi Islands, all of whom were on the Executive Committee of the Tiwi Land Council, and who were present at an environmental forum in Darwin. Kate kindly asked these elders if they had seen hyriids and showed them the relevant illustrations in McKay (2017). On 22 August 2018, Ms Haddden replied that no hyriids were known by these Tiwi people.
- On 9 August 2018, RCW asked Dr Mandy Reid, Collection Manager, Malacology, The Australian Museum, if that institution had any hyriids from the Tiwi Islands in its collection. On 29 August 2018, Dr Reid replied that none were present.
- On 3 September 2018, RCW asked Dr Suzanne Williams, Head of Invertebrate Division, Natural History Museum, London, whom to approach to ask if that institution had any hyriids from the Tiwi Islands in its collection. Dr Williams referred the enquiry to Dr Tom White, presently the Curator responsible for bivalves. Dr White scanned the early registers and the collection for us. On 9 October 2018,

Dr White replied that he was unable to find any relevant specimens. Moreover, he tried various other approaches, such as looking for the names of Tiwi rivers or early settlements (such as Fort Dundas), but to no avail.

- On 20 March 2019, RCW asked Alan Withers, Senior Ranger, Garig Gunak Barlu National Park, Cobourg Peninsula, if he or his wife, Emma, who is an ardent shell collector, if they had seen hyriids and showed them the relevant illustrations in McKay (2017). On 21 March 2019, Mr Withers replied that neither he, nor Emma, had ever seen hyriids on Cobourg Peninsula, despite having spent 19 years there.
- On 6 May 2019, HAJ consulted the OZCAM (Online Zoological Collections of Australian Museums) database, which provides access to records from molluscan collections in Australian museums (CHAFC 2019). The result of this enquiry is that the OZCAM database contains no records of hyriids from the Tiwi Islands or Cobourg Peninsula.
- On 6 May 2019, HAJ consulted the Mussel Project (*MUSSELp*) database, which is a compilation of freshwater mussel collections held by 17 major museums around the world (Graf & Cummings 2018). The result of this enquiry is that the *MUSSELp* database contains no records of hyriids from the Tiwi Islands or Cobourg Peninsula.

Discussion

All our enquiries lead to the conclusion that there are no freshwater mussels (Hyriidae) on either the Tiwi Islands or Cobourg Peninsula (Figure 2). This is surprising as both the islands that comprise the Tiwis are large – Melville having a land area of 5786 km^2 and Bathurst having a land area of 654 km^2. In fact, the former is the second largest island off the Australian mainland after Tasmania (64,419 km^2) and Bathurst is the fifth largest following Groote Eylandt (2285 km^2). As mentioned above, freshwater mussels definitely do live on some islands. Examples of Australian nearshore islands known to support hyriids are (northern) Tasmania (McMichael & Hiscock 1958) and Groote Eylandt (MAGNT P.43248 = 2 specimens collected live from Lake Angurrkburna in September 2009). Examples of islands beyond Australia that also support hyriids are Misool Island (western New Guinea) and the Solomon Islands group (Walker *et al.* 2013). Despite aquatic faunal surveys having been conducted on the Tiwi Islands (Larson 1998; PWCNT 1998; Woinarski & Baker 2002), there are no records of freshwater mussels from the Tiwi-Coburg region. So it seems the absence is genuine and not just the result of inadequate collecting or surveying.

The closest that freshwater mussels exist to the Tiwi Islands is the northern coast of Darwin Harbour (approx. 12.333°S, 130.883°E), which is 43.5 km over water (i.e. Beagle Gulf) to the south. The closest that freshwater mussels exist to Cobourg Peninsula is Oenpelli Billabong (approx. 12.333°S, 133.050°E), which is 95 km over land to the southeast (Figure 1).

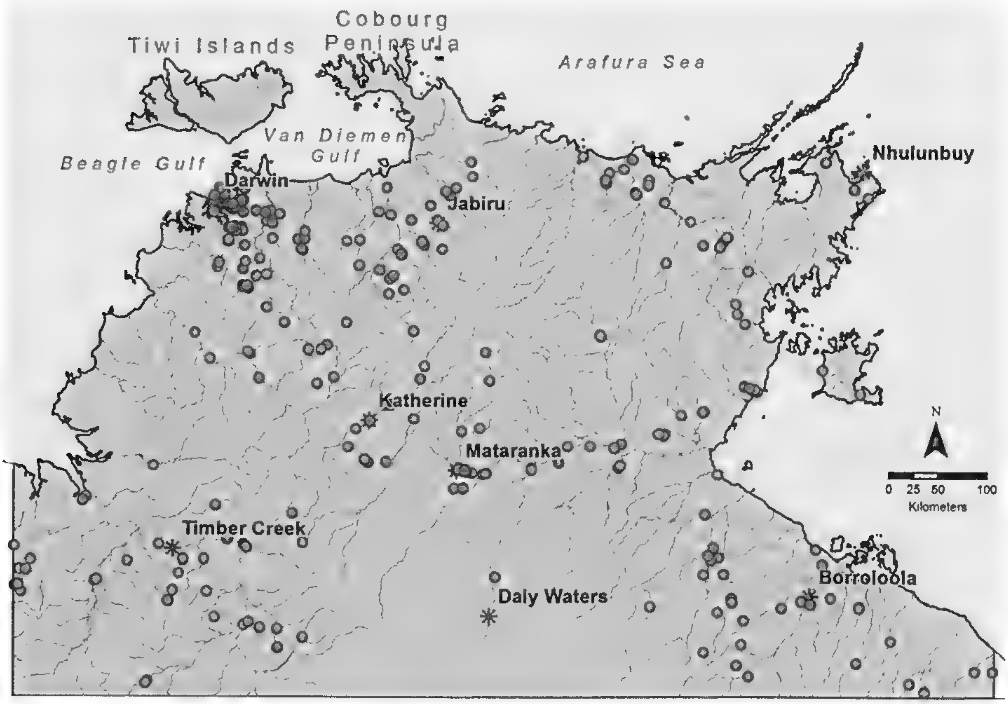

Figure 2. Geographic distribution of freshwater mussels (Hyriidae) for the Top End of the Northern Territory based on all verified specimen lots held by the Museum and Art Gallery of the Northern Territory and the Australian Museum.

Given that the Tiwi Islands (and also Cobourg Peninsula) are theoretically large enough to support freshwater mussels, there are three immediate hypotheses to account for this absence – either the habitats are unsuitable, or there is an absence of fishes to act as suitable host for the mussel's parasitic larva, or there has been significant sea-level change that eliminated them and they have not recolonised.

The Tiwi Islands, particularly Melville Island, have several permanent streams. The major system is the northern drainage, in which are found the largest streams. The southern drainage has shorter, smaller streams (Larson 1998). For Melville Island, the rivers on the northern coast are Kilu-impini Creek, Maxwell Creek, Tjipiri River, Goose (Andranangoo) Creek, Jessi (Aliu) River and Johnston (Tuanungku) River, and (on the southern coast) is Takampirmili Creek. In addition, there are wetlands on both islands, some of which are spring-fed. None of these water systems is reported to be saline.

Salinity is known to be a factor limiting the occurrence of freshwater mussels; Klunzinger *et al.* (2015) showed that *Westralunio carteri* was almost never found at sites where mean salinity was greater than 1.6 gmL^{-1}. This, the only species of hyriid in southwestern Australia, has undergone a 49% reduction in its range in less than 50 years due primarily to secondary (i.e. anthropomorphic) salinisation and its conservation status now qualifies it for listing as Vulnerable under the International Union for the Conservation

of Nature's rankings. However, neither primary nor secondary salinity could currently explain the absence on the Tiwi Islands and Cobourg Peninsula.

Freshwater mussels are unique amongst molluscs in requiring a host to complete larval development. The larva (termed a glochidium) needs to pass through a brief parasitic stage on a fish (Walker 1981b; Smith 1998). The glochidium attaches to the external surface or gills of the host. The host may have an immunity to glochidial attack and absorb the larva. If not, then the glochidium becomes encysted in the epidermis of the host and remains there for one to two weeks, depending on water temperature, undergoing metamorphosis into a juvenile mussel (Humphrey & Simpson 1985). It then excysts and drops to the bottom to develop.

Therefore, it is crucial for a glochidium larva to locate a suitable fish. The most widespread mussel species occurring in the Top End of the Northern Territory, *Velesunio angasi*, is a host generalist and completes its development on at least 19 species of freshwater fishes (Humphrey & Simpson 1985). Though the fish fauna of the Tiwi Islands has received very little scientific study, it would appear to be significantly depauperate compared to the adjacent coast (Larson 1998). In particular, there is apparently only one species of plotosid (eel-tailed catfish) (*Neosilurus ater*), and there are apparently no terapontids (grunters) (Larson 1998); both groups of fishes being widespread, abundant and speciose on the adjacent mainland. The absence of the grunters is highly unusual. The habitats available on the Tiwi Islands would suggest that, at least, the Spangled Perch (*Leiopotherapon unicolor*) and the Sooty Grunter (*Hephaestus fuliginosus*) should be present. The former is ubiquitous elsewhere in the Leichhardtian Fluvifaunal Area, where it inhabits all coastal and inland freshwater creeks and rivers of northern Australia; from the upper Burdekin River in Queensland to the Daly River in the Northern Territory (Larson 1998; Gomon & Bray 2019; Wikipedia 2019; M. Hammer, pers. comm.). Nevertheless, several species of freshwater fish occur in Tiwi Island streams that are suitable hosts for freshwater mussels – melanotaenids (rainbowfishes), atherinids (hardyheads), ambassids (glassfishes), and a number of eleotrids (gudgeons) (Larson 1998; Humphrey & Simpson 1985).

As *Velesunio angasi* is ubiquitous in almost the entire range of freshwater habitats in the Top End (Walker 1998), it would appear that neither a lack of suitable habitat nor a lack of suitable host fishes is responsible for the absence of freshwater mussels on the Tiwi Islands and Cobourg Peninsula. However, it is quite possible that freshwater habitats for hyriids were scarce or absent on the Tiwi Islands around the time of the Last Glacial Maximum. This was a period when the northern Australian climate was significantly drier than at present (Hope 2005). At the beginning of the Holocene, the shift to the modern, monsoon-driven climate coincided with a rapid rise in sea level, isolating the Tiwi Islands and thus preventing colonisation of streams and wetlands on these islands from mainland source populations. Another (less likely) explanation for the absence of mussels on the islands is that for much of the Holocene, sea levels along sections of the northern Australian coastline were somewhat higher than at present (estimates range

from 1–2 m) and they remained higher in the Darwin region until about 2000 years ago (Nott 1996). These elevated sea levels during the Holocene may have drowned out freshwater habitats on the islands. Either a rapid rise in sea level to a considerable height or a gradual rise to a more moderate height would have resulted in seawater incursion into coastal waterways and the increase in salinity would have been lethal to freshwater mussels. Not only would the sea levels have risen, but the temperature of the water would also have risen or fallen considerably as a consequence. Thus, the Tiwi Islands might have been influenced by localised higher water temperature that kept the water levels high (Andrew Rawson pers. comm.).

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Incursion of the bivalve *Potamocorbula faba* into northern Australia: a record from a Holocene archaeological site in Kakadu National Park

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Abstract

This paper reports the discovery of the estuarine bivalve species *Potamocorbula faba* at the archaeological site Ngarradj Warde Djobkeng in Kakadu National Park, Northern Territory. Although presently widespread and abundant in southeastern Asia, this species is not known to be living anywhere in northern Australia today, and neither is it known as a pre-Holocene fossil. Therefore, it must have made a temporary incursion into western Arnhem Land between 3300 and 3600 years ago.

Species profile

Potamocorbula faba is a medium-sized (maximum shell length 16.4 mm) member of the bivalve family Corbulidae (commonly known as basket clams). Its shell valves, which are more equivalve and relatively narrower than many other corbulids, are more or less equilateral (i.e. the beak is located at the centre of the dorsal margin), and are rather elongate – being broadly rounded anteriorly and subrostrate to truncate posteriorly. The ventral margin is more or less straight adding to the narrowness. The exterior is smooth or sculptured with low, concentric ridges, which are strongest marginally. A broadly rounded carina, which is more prominent on the left valve, extends from the beak to the postero-ventral margin. The right valve, which is slightly larger, has a prominent cardinal tooth in front of the sunken ligament at the beak internally. The left valve has a small anterior cardinal tooth and a large trigonal chondrophore with a median cleft containing a narrow ligamentary extension. The pallial line has the merest indication of a sinus posteriorly. The shell itself is pure white. It is polished externally and covered by a thin, glossy, brownish grey periostracum when live. The posterior quarter is often covered with a reddish brown mineralogical deposit which precipitates on it when it is in its normal life posture (i.e. buried vertically into the substrate with the posterior end uppermost). The retractile siphons consist of separate inhalant and exhalant tubes, both extremely short; the inhalant siphon is larger and brown and has a fringe of relatively long tentacles, whereas the exhalant siphon is smaller and pinkish with a fringe of relatively short tentacles. The labial palps are smaller than the gills.

Potamocorbula faba is a suspension feeder using the gills to sort particulate food out of the incoming seawater. It burrows shallowly (0–3 cm) in sandy mud in the intertidal and shallow subtidal zones in estuaries in southeastern Asia (Ambarwati & Trijoko 2011; Hariyadi *et al.* 2017). In fact, it is one of the most abundant bivalves in estuaries in Indonesia, with densities between 3549–10,000 individuals m^2 in the Kepetingan and Porong River estuaries (Ambarwati & Trijoko 2011). *Potamocorbula* species in general are highly tolerant of low oxygen concentrations and euthrophic environments (Thompson & Parchaso 2012), and *P. faba* is no exception as it can live in highly polluted situations (Ambarwati & Trijoko 2011: 168).

Prior to this report, *Potamocorbula faba* has never been recorded from Australia.

The taxonomy and nomenclature of corbulids in general are poorly known because the family has little commercial importance. We are aware that Huber (2010: 771) cursorily dismissed the specific name *faba* as "preoccupied" in *Corbula* but no replacement name has ever been proposed and, in fact, there is no preoccupation when *faba* is placed in the genus *Potamocorbula*. Moreover, Huber (2010; repeated in MolluscaBase 2019) considered *P. faba* synonymous with *P. fasciata*. We reject that synonymy, which comes with the supposition that *P. fasciata* is a "highly variable species in coloring and somewhat less in elongated shape" (Huber 2010). To us they are completely different species with the 'true' *P. fasciata* having a more rostrate spoon-shaped posterior extremity, a more convex ventral margin, a pallial line is relatively further from the margin of the valve and, most importantly, several broad reddish rays emanating from the beak (pers. obs.). Illustrations of 'typical' *P. faba* are given by Huber (2010: 470) from "Thailand-Vietnam" and by Poppe (2011: 386) from the Philippines. Significantly, the specimen illustrated by Poppe (2011) was collected at 20–22 m, which is considerably deeper than that known for *P. faba*. This corollary of our decision that *P. faba* and *P. fasciata* are different species is that we do not know the geographical distribution of either of them, other than both occur broadly in southeastern Asia.

Archaeological introduction

Ngarradj Warde Djobkeng is a rockshelter located within Mirarr Country on the northwestern margin of the Arnhem Land Plateau of the Northern Territory near the East Alligator River. The site was first recorded and excavated by Harry Allen in 1972 as part of the Alligator Rivers Environmental Fact-Finding Study, an initiative put in place to assess the archaeological potential of the soon-to-be Kakadu National Park (Kamminga & Allen 1973). Allen's 1972 test pit produced a rich archaeological assemblage and he returned in 1977 to conduct further excavations. The 1977 field season opened up a further 12 m^2 at the site, bringing the total area excavated to 13 m^2.

Archaeological material at Ngarradj Warde Djobkeng extends to a depth of 185 cm, and initial occupation of the site has been tentatively placed at 18,000–26,000 BP (Allen & Barton 1989). This report, however, focusses on finds recovered from the dense midden located in the upper Holocene aged layers of the site. The Ngarradj Warde

Djobkeng midden stretches across the sheltered portion of the rockshelter, beginning approximately 15 cm below the surface of the site, and extending to a depth of approximately 65 cm (Allen & Barton 1989). It is comprised primarily of faunal remains from estuarine environments and is heavily dominated by molluscs. Radiocarbon dates for the midden layers indicate that it was deposited rapidly over a 1100-year period between 4400 and 3300 cal BP. The six *Potamocorbula faba* specimens discussed in this report were located in the upper portion of the midden, towards the back of the shelter, in layers which ranged in depth from 10–25 cm. Radiocarbon dates for these layers suggests that these specimens were deposited between 3600 and 3300 cal BP.

The specimens from Ngarradj Warde Djobkeng

Six specimens of *Potamocorbula faba*, all single valves, were recovered from the midden at Ngarradj Warde Djobkeng (Table 1). These specimens are currently housed in the Archaeology Store at the Museum and Art Gallery of the Northern Territory (MAGNT; collection registration number AA 1997.002). The specimens within the Ngarradj Warde Djobkeng collection do not have individual numbers, so we have designated numbers 1–6 for them in Table 1 so they can be referred to individually.

One specimen (number 1) was collected during the 1972 phase of the excavation and five specimens (numbers 2–6) were collected during the 1977 phase.

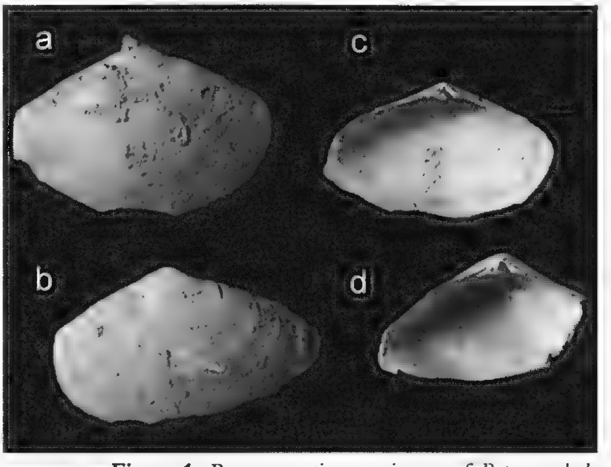

Figure 1. Representative specimens of *Potamocorbula faba* excavated from Ngarradj Warde Djobkeng (Kakadu National Park, Northern Territory), Museum and Art Gallery of the Northern Territory, Archaeology collection AA 1997.002. Details of shell form, full sample number, and full dimensions of all the specimens are given in Table 1. **a,** exterior of specimen no. 6, length 12.2 mm; **b,** exterior of specimen no. 5, length 12.7 mm; **c,** interior of specimen no. 3, length 10.8 mm; **d,** interior of specimen no. 2, length 9.8 mm. (MAGNT)

Table 1. Data for all the specimens of *Potamocorbula faba* excavated from Ngarradj Warde Djobkeng (Kakadu National Park, Northern Territory), Museum and Art Gallery of the Northern Territory, Archaeology collection AA 1997.002.

Specimen no. (and form)	Sample no.	Dimensions - max. shell length x width (mm)	Comment
1 (single right valve)	NWD'72 B1 Spit 3	10.3 x 7.6	Posterior end missing completely
2 (single left valve)	NWD'77/1 SQ1 D layer 2	9.8 x 5.9	Anterior and ventral margins broken
3 (single left valve)	NWD'77/1 SQ1 E4 (west)	10.8 x 6.3	Ventral margin broken
4 (single right valve)	NWD'77/1 SQ1 E5 (north)	12.5 x 7.8	Postero-ventral margin broken
5 (single left valve)	NWD'77/1 SQ1 G-K6	12.7 x 7.3	Ventral margin broken
6 (single right valve)	NWD'77/1 SQ1 G-K6	12.2 x 7.7	Ventral margin broken

The longest specimen (number 5) (Figure 1b) has a maximum shell length of 12.7 mm. The most intact specimen is number 3 (Figure 1c). The smallest specimen (number 2) (Figure 1d) has a maximum shell length of 9.8 mm, but this specimen has extensively broken anterior and ventral margins so it would have been larger in life. Indeed, all the specimens are broken to some degree (Figure 1). From (a) the smoothness of the interior of each shell valve, (b) the intactness of the hinge teeth and chondrophore (which would have been abraded and/or broken off soon after death), and (c) the complete lack of encrusting organisms (animals like barnacles, tubeworms and bryozoans colonise shells very soon after the death of the mollusc that formed them), we conclude that they were all collected live and the damage now evident on them occurred as they were being opened by the gatherers. No delamination, or micro-cracks, or charcoal remains are present on any of the shell valves suggesting they were not directly fired to get them open (though they could have been boiled in water).

Discussion

The layers in which these specimens were found date to a period when environmental conditions in the region of the East Alligator River were significantly different to those in the area today. Rising sea levels initiated by shifting climatic conditions during the transition from the Pleistocene to the Holocene triggered a series of substantial environmental and geomorphological changes across the entire Alligator Rivers Region (Woodroffe *et al.* 1985). These rising sea levels flooded the low-lying continental shelf which connected Australia to New Guinea and resulted in a marine intrusion into the incised river valleys across the northern portion of the continent. This marine influence resulted in the development of extensive tidal flats and brackish water swamps along these river systems which were gradually colonised by mangrove forests (Woodroffe *et al.* 1988). By 6800 BP, extensive mangrove forests were present along the river systems in this region (Woodroffe 1988). The stabilisation of sea levels around 6000 BP was accompanied by an increased level of sedimentation in these river systems (Woodroffe *et al.* 1985). Unable to keep up with this increase in sediment, these extensive mangrove forests began to decline and were gradually replaced by freshwater sedges and grasses (Clarke & Guppy 1988). In some areas of the region, sediment levees began to accumulate around the mangrove forests during this period, leading to the development of hypersaline swamps/floodplains flanking the river system (Hope *et al.* 1985).

The specimens described here were deposited during this final transition period, as the region moved from brackish water swamps to freshwater wetlands. The landscape surrounding the site of Ngarradj Warde Djobkeng during this time would have been a mosaic environment. Stands of mangroves, and brackish water swamp-like conditions would have still been present along the rivers in the region, however, newly formed large permanent bodies of freshwater dominated by grasses and sedges would have also been present around the site.

Conclusion

It is probable that Aboriginal people around the East Alligator River gathered *Potamocorbula faba* for food, as is done presently by people in Indonesia (who often use this species for the commercial traditional dish called *lontong kupang*, the raw ingredient of *krupul* and *petis*, as well as for animal food (Ambarwati & Trijoko 2011; Hariyadi *et al.* 2017)).

Middens formed by traditional gatherers of shellfish are particularly good sources for recording shifts in distributions of molluscs beyond their 'natural range'. Good examples of this are the southern New Zealand limpet *Cellana denticulata* that had an incursion into northeastern New Zealand during the Holocene (Willan 1974) and the eastern African nerite *Nerita textilis* that had an incursion into eastern Indonesia during the Pleistocene (Eichhorst & Szabo 2004); it seems that sometime between 35,000 years ago and the end of the Holocene, 10,000 years ago, the latter vanished from Indonesian waters, or at least became too rare to turn up in middens (Eichhorst 2016: 577).

The occurrence of *Potamocorbula faba* in western Arnhem Land is significant because this represents the first example of a mollusc that had a natural incursion into Australia (in this case, presumably from the southern coast of New Guinea) in historical times, but then failed to persist. It is certain that *P. faba* is not living in Arnhem Land, or anywhere else in northern Australia, today. The size of the temporary population will never be known, but its demise is probably related to the change from brackish water swamps to freshwater wetlands through the Holocene period.

Acknowledgements

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Catch, discard and bycatch rates in the Western Gulf of Carpentaria Mud Crab Fishery: summary for 2017 and 2018

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Abstract

Catch, discard and bycatch rates in wire mesh mud crab pots set in two regions of the Western Gulf of Carpentaria are summarised for 47 fishery observer trips undertaken aboard commercial crab fishing vessels in the months of April and May in 2017 and 2018. Catch rates of the Giant Mud Crab (*Scylla serrata*) in both regions and both years were relatively high (i.e. 0.74–1.02 kg/pot-lift), as were discard rates (i.e. 59–69% of individual Mud Crabs captured). Between 33% and 57% of all discarded crabs had attained the minimum legal size, but were released on the basis that they were 'commercially unsuitable', having a soft shell, low vigour and low meat content. A total of 88 individuals from 12 different taxa were recorded as bycatch from 4337 pot-lifts. Catfishes (*Arius* spp.) were by far the most numerous bycatch, accounting for 64% of all individuals. There were no interactions between mud crab pots and threatened, endangered or protected species during any observer trips. The low incidence of bycatch in wire mesh mud crab pots is consistent with previous observations on the use of this type of gear both locally and interstate.

Introduction

Commercial mud crab fishery licensees in the Northern Territory, Australia, use baited wire mesh pots (approximately 60 x 70 x 20 cm; L x W x H) to capture mud crabs (*Scylla* spp.) in mangrove and mud-flat habitats. One of the primary indicators of the success of crabbing operations (and mud crab abundance) is catch rate, typically expressed as the quantity of crabs retained (in kg) per unit of fishing effort (as pot-lifts).

Controls on the commercial harvest of mud crabs in the Northern Territory include a Minimum Legal Size (MLS) for each sex (i.e. 140 mm carapace width [CW] for males and 150 mm CW for females) and a prohibition on the take of recently moulted "Commercially Unsuitable Crabs" that exceed the MLS, but have a soft shell, low vigour and low meat content; the latter being an industry initiative to reduce transport mortality and increase the market price. These regulations often mean that a large proportion of the mud crab catch (i.e. >70%; Ward *et al.* 2008) has to be returned to the water, with such individuals referred to here as 'discards'. The discard rate can vary in both space

and time, depending on the size structure of the local population and the timing of moulting events.

The bait used in mud crab pots also attracts a range of scavengers and predators to the gear, some of which get trapped and become 'bycatch'. There are relatively few accounts of bycatch in Australian mud crab fisheries, but those that are available (e.g. Butcher *et al.* 2012; Grubert & Lee 2013) suggest that bycatch in rigid wire mesh pots is considerably lower than that in collapsible trawl mesh pots, primarily because of the smaller mesh used in the latter gear type.

Management of the Northern Territory Mud Crab Fishery underwent significant reform in 2017, with two discrete regional management units implemented – the Arafura-West Mud Crab Fishery, which encompasses the Northern Territory coastline anti-clockwise from Cape Grey (13°0' S, 136°39' E), and the Western Gulf of Carpentaria Mud Crab Fishery which constitutes the remainder of the Northern Territory coast to the Queensland border (Figure 1).

The performance of these management units is assessed each year through an analysis of catch rates over the indicator months of April and May, derived from compulsory fishery logbooks. The catch rate during this period is considered a reliable indicator of that for the remainder of the year. One of three seasonal closures (beginning in October) may then be imposed if the April–May catch rate is particularly low. A fishery

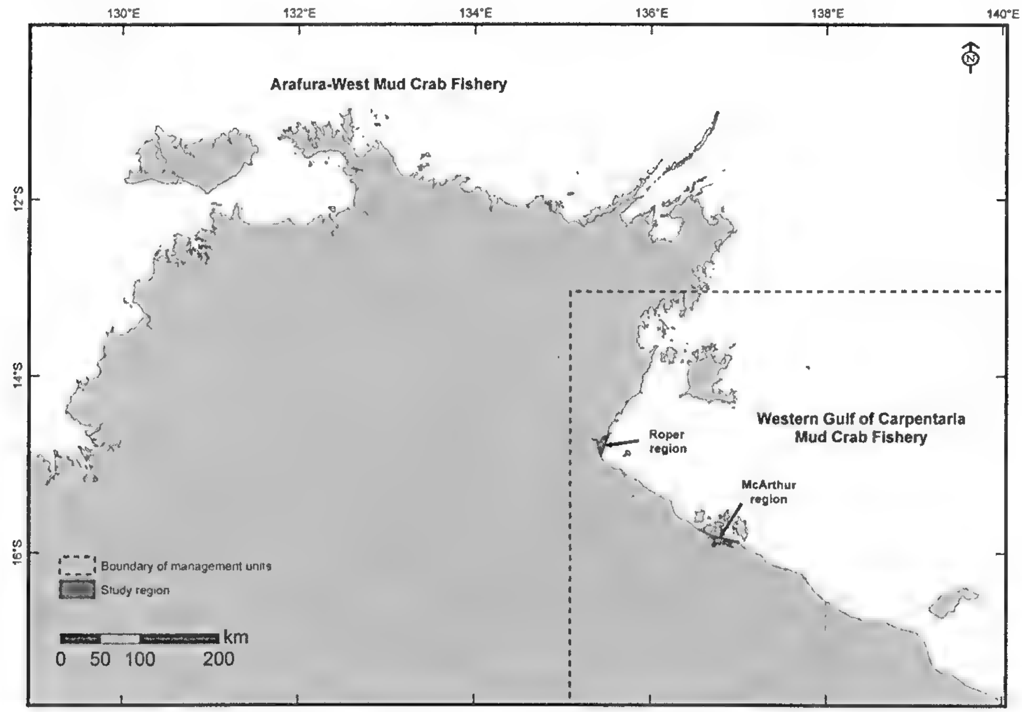

Figure 1. Extent of mud crab fishery management units and study regions.

observer program targeting the same months is also undertaken to validate reported data and collect information on discard rates and bycatch composition, which is not typically reported by licensees.

Although some observer trips have been undertaken in the Arafura-West Mud Crab Fishery, the primary focus to date has been the Western Gulf of Carpentaria Mud Crab Fishery, as this fishery has produced around 70% of the total commercial mud crab catch in the NT over the last two decades (Northern Territory Government 2018). This being the case, the information provided here is confined to observations of the catch, discard and bycatch rates in only the Western Gulf of Carpentaria Mud Crab Fishery.

Materials and Methods

Fishery observers recorded catch, discard and bycatch information during fishing operations by several commercial mud fishery licensees across the months of April and May 2017 and 2018. Observations centred on two fishing regions within the Western Gulf of Carpentaria Mud Crab Fishery – the 'Roper region' extending from Warrakunta Point (14°39' S, 135°32' E) in the north to the Towns River in the south (14°55' S, 135°26' E), and the 'McArthur region' extending from Carrington Channel (15°46' S, 136°36' E) in the west to Pelican Spit in the east (15°52' S, 137°0' E) (Figure 1).

Observers typically joined well established licensees with at least 10 years' experience in the industry. The number and timing of observer trips was dictated by staff availability and the willingness of licensees to allow them on board. Given these logistic constraints, no attempt was made to concentrate on a particular stage of the lunar cycle.

In most cases, only one observer trip was conducted per day, generally on the larger of the two daily high tides. However, there were a few instances where licensees checked their pots twice a day and were joined by observers on both occasions. These events were treated as two separate observer trips. Trips usually lasted 3–6 hours depending on the number of pots checked (which typically ranged from 60–120) and their proximity to the licensee's camp.

Results

Observers witnessed in excess of 800 pot-lifts (by two to five different licensees) in each region and both years of the study (Table 1). The mud crab catch consisted entirely of the Giant Mud Crab (*Scylla serrata*), the only species of *Scylla* present in the region. Regional catch rates were relatively high (i.e. 0.74–1.02 kg/pot-lift), as were discard rates (i.e. 59–69% of individuals captured). Overall catch rates (for both regions combined) were 0.91 kg/pot-lift in 2017 and 0.78 kg/pot-lift in 2018.

Most discarded crabs (i.e. >60%) were males, with the ratio of under-sized male to soft-shelled male crabs being broadly similar in several cases (Table 2). The proportion of under-sized female crabs was always higher than that of soft-shelled female crabs, with the greatest difference between these categories observed in the Roper region.

Table 1. Mud crab catch, volumetric catch rate and numeric discard fraction for observer trips undertaken in each region and year.

Location	Year	Observer trips	Licensee count	Retained catch (kg)	Effort (pot-lifts)	Catch rate (kg/pot-lift)	Total catch (number)	Discard fraction
McArthur region	2017	10	4	926.64	885	1.05	3489	68%
	2018	13	5	832.12	1131	0.74	3534	69%
Roper region	2017	11	2	1014.42	1254	0.81	2234	63%
	2018	13	3	889.67	1067	0.83	2782	59%

Table 2. Proportion of mud crabs in each discard category (relative to total discards) by region and year.

Location	Year	Under-sized		Soft-shelled	
		Male	Female	Male	Female
McArthur region	2017	29%	14%	49%	8%
	2018	30%	26%	34%	10%
Roper region	2017	37%	22%	37%	4%
	2018	46%	22%	27%	5%

A total of 88 individuals from 12 different taxa were recorded as bycatch from all observed pot-lifts (n = 4337). Fewer bycatch taxa were encountered in the McArthur region (Table 3) than the Roper region (Table 4). Catfishes (*Arius* spp.) were the most common bycatch in both regions and years. Of the other bycatch taxa observed, fewer than 10 examples of each were recorded from a given region during the study period.

No interactions between wire mesh crab pots and threatened, endangered or protected species (e.g. crocodiles, sawfish or turtles) were detected during any of the observer trips.

Discussion

The overall observed catch rates (for both regions combined) in 2017 and 2018 were 0.91 kg/pot-lift and 0.78 kg/pot-lift, respectively. These values were 21% above the reported catch rate for the Western Gulf of Carpentaria Mud Crab Fishery in both years (i.e. 0.75 kg/pot-lift and 0.65 kg/pot-lift, respectively; Northern Territory Government, unpublished data). This was not unexpected given that observations focused on the activities of more experienced (and efficient) licensees.

Discard rates ranged from 59–69% depending on year and region, and fell within the range of values previously reported for wire mesh crab pots set in the Northern Territory (Ward *et al.* 2008; Grubert & Lee 2013). However, it should be noted that previous studies did not distinguish discards as either under-sized or soft-shelled as was done here (Table 2).

The survival rate of discards is not known, but is assumed to be relatively high, given observations of repetitive recaptures of tagged crabs over the period of a few days

Table 3. Annual bycatch totals and pooled bycatch rate* for mud crab pots set in the McArthur region.

Class	Species	2017	2018	Pooled bycatchrate/1000 pot-lifts
Actinopterygii	Bream (*Acanthopagrus* spp.)	1	1	1.0
	Catfish (*Arius* spp.)	2	21	11.4
	Goldspotted Rockcod (*Epinephelus coioides*)	0	3	1.5
	Conger Eel (family Muraenidae)	1	0	0.5
	Spotted Scat (*Scatophagus argus*)	0	1	0.5
Chondrichthyes	Milk Shark (*Rhizoprionodon acutus*)	1	1	1.0
Malacostraca	Blue Swimmer Crab (*Portunus* spp.)	0	6	3.0

* Pooled bycatch rate calculated as the sum of the annual bycatch totals divided by the sum of annual pot-lifts (i.e. 2016 for the McArthur region) multiplied by 1000.

Table 4. Annual bycatch totals and pooled bycatch rate* for mud crab pots set in the Roper region.

Class	Species	2017	2018	Pooled bycatch rate/1000 pot-lifts
Actinopterygii	Barramundi (*Lates calcarifer*)	0	1	0.4
	Barred Javelin (*Pomadasys kaakan*)	1	0	0.4
	Blue Threadfin (*Eleutheronema tetradactylum*)	2	0	0.9
	Bream (*Acanthopagrus* spp.)	0	2	0.9
	Catfish (*Arius* spp.)	19	14	14.2
	Goldspotted Rockcod (*Epinephelus coioides*)	0	1	0.4
	Mullet (*Mugil* spp.)	2	0	0.9
	Tripletail (*Lobotes surinamensis*)	2	0	0.9
Chondrichthyes	Milk Shark (*Rhizoprionodon acutus*)	5	0	2.2
Malacostraca	Blue Swimmer Crab (*Portunus* spp.)	1	0	0.4

* Pooled bycatch rate calculated as the sum of the annual bycatch totals divided by the sum of annual pot-lifts (i.e. 2321 for the Roper region) multiplied by 1000.

(Grubert & Lee 2013). The high proportion of discarded male crabs was not surprising because male crabs are known to dominate the catch during the first half of the year (Knuckey 1999).

Soft-shelled male crabs constituted the largest, or second largest, proportion of all discards across the months of April and May in both regions and years. By contrast, soft-shelled female crabs formed the smallest percentage of discards in all cases. This observation is consistent with the reproductive cycle of *Scylla serrata*, where mating activity peaks towards the end of the year (Knuckey 1999). Mating can only take place when the shell of the male crab is hard and that of the female crab is soft. This necessitates that the moult cycles of the sexes are asynchronous (at least four adults), with males generally moulting earlier in the year and females later in the year.

The mandatory use of escape vents in commercial mud crab pots in the Northern Territory (implemented in June 2018) is expected to reduce the discard rate in the fishery by allowing more under-sized crabs to escape from pots. While future fishery observer work during the focal period (i.e. April–May) might detect a reduction in the discard rate, it may not capture the full impact of this legislative change, as observations are restricted to a two-month window. Expansion of the fishery observer program beyond these months is unlikely (because of costs), but a more comprehensive assessment of discard rates could be undertaken by industry voluntarily.

The low incidence of bycatch in wire mesh mud crab pots described here is consistent with previous reports on the use of this gear in both the Northern Territory (Hay *et al.* 2005; Grubert & Lee 2013) and New South Wales (Butcher *et al.* 2012). Of the 12 taxa recorded as bycatch during the current study, three were new records for wire mesh pots in the Northern Territory – the Spotted Scat (*Scatophagus argus*), the Tripletail (*Lobotes surinamensis*) and an undifferentiated species of conger eel (family Muraenidae). The number of taxa recorded as bycatch in the Roper region was greater than that in the McArthur region. However, this is of limited significance because a large proportion of the taxa recorded in both regions were represented by just one individual.

Similar to Hay *et al.* (2005), who observed a total of 10,416 pot-lifts over a two-year period, there were no observations of interactions between wire mesh crab pots and threatened, endangered and protected species during the current study.

Acknowledgements

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A record of fish predation by the mangrove crab *Epixanthus dentatus* (Brachyura: Oziidae)

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Abstract

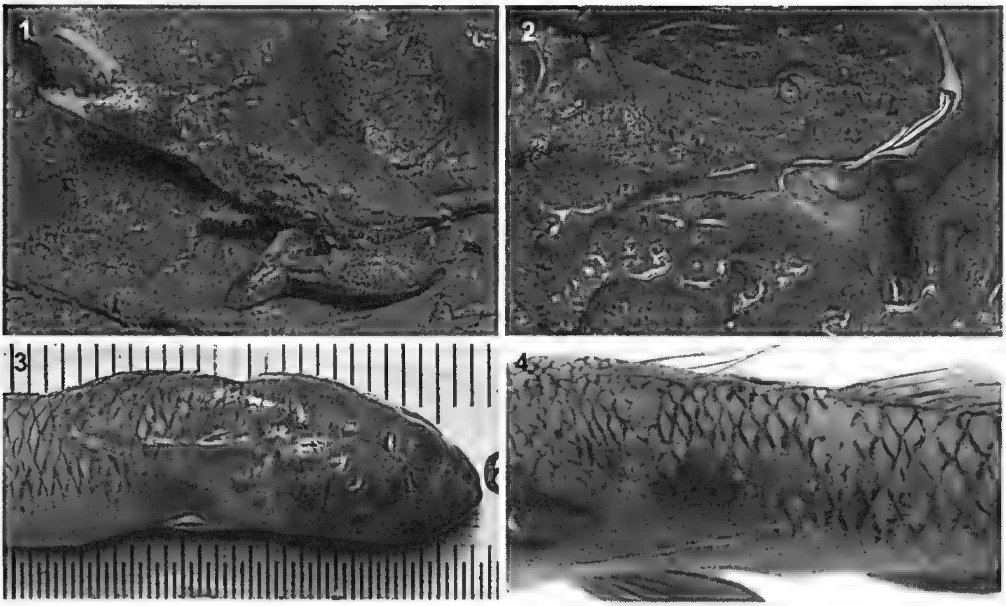
A case of predation by the oziid crab *Epixanthus dentatus* on the gobiid fish *Acentrogobius viridipunctatus* within a mangrove forest in Darwin Harbour, Australia, is reported here. The observation was made after the moment of prey capture, however both the patterns of injury and freshness of the carcass indicate the goby had been ambushed live during the previous ebb tide. This is the first record of *E. dentatus* predating fish and increases the known diet of the species.

Introduction

Field observations show that the oziid mangrove crab *Epixanthus dentatus* uses a sit-and-wait ambush technique to capture a variety of slow-moving mangrove invertebrates (Cannicci *et al.* 1998; Dahdouh-Guebas *et al.* 1999; Vannini *et al.* 2001). Investigations on its natural diet indicate that although *E. dentatus* is capable of capturing most resident invertebrate faunas, it preys primarily on other brachyurans and shelled gastropods (Cannicci *et al.* 1998; Vannini *et al.* 2001). Interestingly, despite the resourcefulness of *E. dentatus* as a versatile epibenthic predator there are presently no published accounts of it preying on mangrove-dwelling fishes. Previous work on stomach content analysis (Dahdouh-Guebas *et al.* 1999) and examinations of prey remains accumulated at home dens (Vannini *et al.* 2001) both failed to record fish in the diet of *E. dentatus*. This note reports *E. dentatus* with a freshly captured individual of the Greenspotted Goby (*Acentrogobius viridipunctatus*). This is, to my knowledge, the first record of *E. dentatus* preying on fish.

Materials and methods

The observation occurred on 17 February 2018, approximately 4.5 hr after a 6.7 m high spring tide in a seaward fringing *Rhizophora* mangrove forest, 260 m northwest of the East Arm Boat Ramp (12° 28.917'S, 130° 54.601'E), Darwin Harbour, Australia. Environmental conditions, as recorded at 09:00 hr were 26.4°C mean air temperature, 91% relative humidity, and 32.0°C sea temperature (Bureau of Meteorology 2018). To determine the relative age of the goby's carcass, an adapted version of the Torry Freshness Score (Shewan *et al.* 1953) for whole fish was employed. Unfortunately, no preserving alcohol was available at the time of collection. As such, the carcass was transported to the laboratory in seawater and subsequently persevered in 70% ethanol.



Figures 1–4. Predation of the Greenspotted Goby (*Acentrogobius janthinopterus*) by *Epixanthus dentatus*. 1–2. *E. dentatus* clutching the dead *A. janthinopterus* in its minor cheliped. 3. Freshly dead *A. janthinopterus* displaying opalescent body colouration and a semi-translucent eye. 4. Magnified view of damage to the goby's trunk caused by the crab's two chelipeds. (Adam Bourke)

Identification of the goby was done by the Museum and Art Gallery of the Northern Territory, and the specimen is deposited in the fish collection there under the registration number S.18180-001.

Observations

An adult female *Epixanthus dentatus* was observed in ambush position against the stilt roots of a *Rhizophora stylosa* tree clutching a dead *Acentrogobius viridipunctatus* (52.7 mm total length) in its minor cheliped (Figures 1, 2). To undertake an assessment of the state of the goby's carcass, the crab was caught and the fish carefully removed from its claw using a small stick. An *in situ* examination of the carcass suggested the goby was only recently dead. Characteristics indicating 'freshness', according to Shewan *et al.* (1953), included opalescent body colouration, clear body slime, red gill filaments and convex, semi-translucent eyes (Figure 3). Further examination under a dissecting microscope revealed that the body damage was consistent with seizure and crushing from the crab's two chelipeds. Patterns of injury comprised two lacerated and compressed regions with puncture marks, membranous tears to the dorsal fins and caudal fin, and gross loss of scales around the injured areas (Figure 4).

Discussion

Predation of fish is well known amongst highly mobile brachyurans such as the Portunidae (e.g. Hill 1976; Paul 1981; Choy 1986; Baeta *et al.* 2006), but is currently unreported in

Epixanthus dentatus, which is a member of the much less active family Oziidae. Given the ability of *E. dentatus* to capture prey, not only on the mud surface but also above and below the water surface (Vannini *et al.* 2001), it is not surprising this species has the capacity to opportunistically ambush active prey such as mangrove-dwelling gobies. However, the observation presented here was made after the moment of prey capture and *E. dentatus* has been recorded scavenging vertebrate carrion (Cannicci *et al.* 1998). As such, the possibility that the goby was scavenged cannot be ruled out. Nonetheless, given the patterns of injury on the fish's body and the freshness of the carcass under such high environmental temperatures, the most likely scenario was that the goby was ambushed during the previous ebb tide. Capture of this goby widens the known diet of *E. dentatus* to now include small fish.

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First record of the Grass Cross Spider (*Argiope catenulata*) of the family Araneidae in Australia

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Abstract

The Grass Cross Spider, *Argiope catenulata* (Doleschall 1859), is newly recorded from Australia based on collections made in early 2019 from three dams in the vicinity of Darwin, Northern Territory. Both sexes are illustrated, as is the egg sac and habitat. Preliminary discussion is given about its introduction and dispersal.

Introduction

Orb-weavers of the genus *Argiope* (family Araneidae), known in Australia as St. Andrew's Cross Spiders, are diurnal spiders known for the bold pattern of the females. Except for adult males, members of this genus construct circular webs distinctive because of the presence of a white silk stabilimentum which generally takes on a cross shape in most species. The legs are held in pairs in line with the stabilimentum, and the females have distinctive white or silver setae on the carapace. In the subfamily Argiopinae, the posterior eye row is procurved (i.e. the lateral eyes in one of the two transverse rows of eyes are more anterior than the median eyes), resulting in the lateral eyes being closely paired and the median eyes in a loose central square. The genus *Argiope* is further distinguished by the placement of the posterior median eyes closer to each other than to the lateral eyes. Males are significantly smaller and duller than females (Levi 1983).

Members of the genus occur throughout the world, especially in tropical and subtropical regions (World Spider Catalog 2019). The most recent review by Levi (1983) of the Western Pacific region listed 16 *Argiope* species for Australia, eight of which are present in the Northern Territory (i.e. *A. aetherea*, *A. dietrichae*, *A. katherina*, *A. ocyaloides*, *A. picta*, *A. protensa*, *A. radon* and *A. trifasciata*). Herein, we add an additional species, *A. catenulata*, thus bringing the total number of species recorded from Australia to 17.

Records and identification

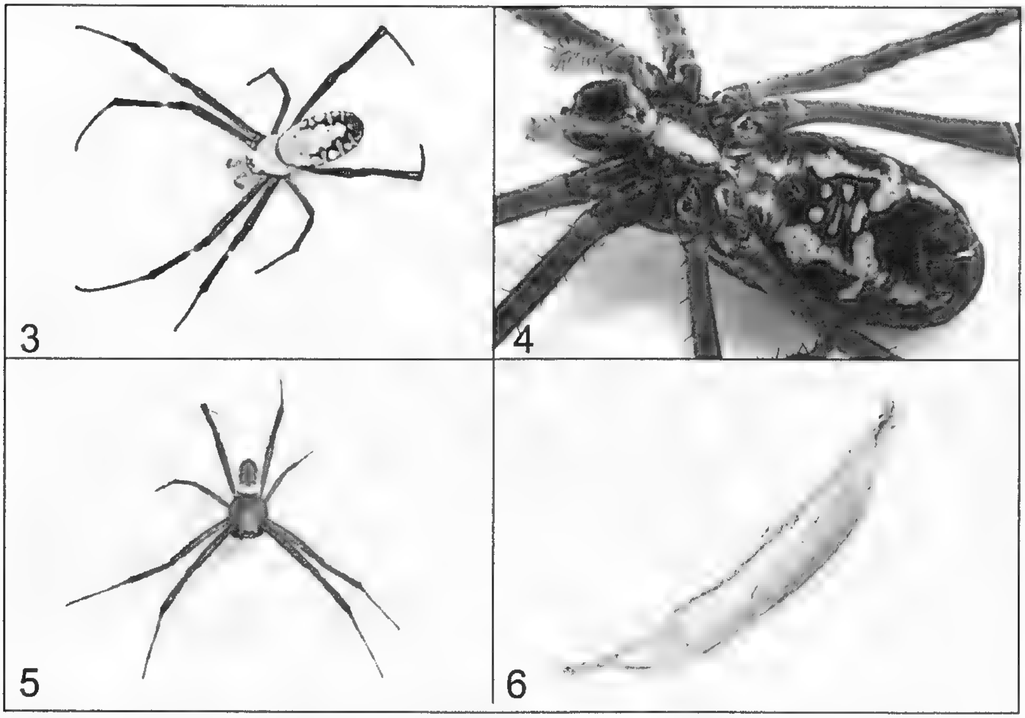
On 12 February 2019, the authors were taken by airboat onto Fogg Dam by Parks and Wildlife Ranger Dave McLachlan. Once out on the water, we spotted several individuals

Figure 1. Sedges at the water's edge, Fogg Dam. (Caitlin Henderson)

of an unidentified species of *Argiope* in orb webs stretching between the stems of the grass-like sedge *Eleocharis dulcis* (known as Chinese Water Chestnut) (Figure 1). Two specimens of these spiders were collected, which we subsequently identified using keys, descriptions and illustrations (including that of the epigynum) from Levi (1983), as *Argiope catenulata*, known commonly as the Grass Cross Spider.

Figure 2. Adult female Grass Cross Spider (*Argiope catenulata*) feeding in her web. (Caitlin Henderson)

Argiope catenulata (Figures 2–5) has distinct identifying markings on both the dorsal and ventral sides of the opisthosoma (i.e. the posterior part of the body, which includes the respiratory organs and the heart, behind the prosoma) in the female. The dorsal pattern could be interpreted as a silver trident, with the outer tines broken by four pairs of orange/yellow spots or transverse lines. The cephalothorax is silver (Figure 3). The male is also more readily identifiable than some other species by the margin of bright silver/white hairs on the dorsal opisthosoma



Figures 3–6. Adults and egg sac of *Argiope catenulata*: **3.** Adult female, dorsal view. **4.** Adult female, ventral view. **5.** Adult male, dorsal view. **6.** Egg sac detached from silk line. Lengths: female = 18 mm; male = 5 mm; egg sac = 45 mm.

(Figure 5). Most distinctive is the female venter where two yellow/white transverse bands meet, or almost meet, in the centre (Figure 4). Particularly, these features distinguish it from the Painted St. Andrew's Cross Spider (*Argiope picta*). The egg sacs are elongated and pale yellow-brown, hanging from the centre of a strong silk line in vegetation near the web (Figure 6).

Further investigation revealed that *A. catenulata* was first photographed at Fogg Dam in 2014 by Kim McLachlan. These observations were also made during an airboat trip. However, the photographs were not forwarded to specialists for identification.

A subsequent trip on 19 February 2019 to Harrison Dam (Figure 7) (a little over 3 km from Fogg Dam) turned up only a single specimen of *A. catenulata* despite us searching a large portion of the near-identical habitat. Notably, this specimen was found at a site that is frequently used for launching boats. Incidentally, we noted that *A. picta* was also present around the edges of Harrison and Fogg Dams.

On 28 February 2019, a small population of *A. catenulata* was also found by us at Manton Dam (Figure 7), and a single female was also spotted along the drainage lines between ponds at Durack, Palmerston (Figure 7). Similar habitat was searched in Berry Springs, but the species was not found to be present.

Specimens of *Argiope catenulata* from these collections are deposited in the Natural Science collection at the Museum and Art Gallery of the Northern Territory under the following registration numbers: A.5220–A.5224, A.5228 (= 6 lots containing 7 specimens in total from Fogg Dam); A.5227 (= 1 lot containing 1 specimen from Harrison Dam); A.5226, A.5227 (= 2 lots containing 2 specimens in total from Manton Dam).

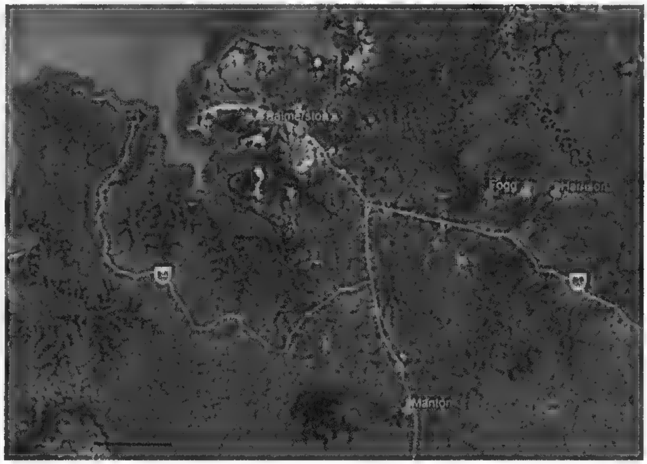


Figure 7. Map of Darwin and rural area pinpointing localities at which *Argiope catenulata* was found in February 2019.

Discussion

While the same Parks and Wildlife airboat is in use across all three dams where the spiders were found, it is not in use at the ponds at Palmerston. However, reed-cutting equipment from Palmerston has been in use at Fogg Dam, so this is one possible explanation for this distribution pattern. Ballooning is the natural dispersal method of *Argiope* species, but patchy access to permanent water bodies in this region may have limited the success of hatchlings making it to further suitable habitat. Adult females taken from the sites at which the specimens were observed did not thrive in captive conditions suitable for other local *Argiope*, suggesting that these spiders are unable to survive away from water.

Argiope catenulata is a distinctive spider common to rice paddies (Poolprasert & Jongjitvimol 2014) from India to the Philippines, down to the southern coast of New Guinea (Levi 1983), and including the Malaysian Peninsula (Tan 2018) and Indonesia (Cheng & Kuntner 2014). It has not been previously recorded from Australia. On Fogg Dam, it is now a common species most frequently occurring in stands of *Eleocharis dulcis,* but also in limited numbers on other tall sedges (Figure 1). It appears largely restricted to the water, not being found on land or with land-adjacent webs.

The present abundance of *A. catenulata* at Fogg Dam raises questions about its relationship with this particular site. While it is not impossible that *A. catenulata* is native to northern Australia, given the proximity to New Guinea, the history of the dam itself is of note. In the 1950s, Fogg Dam was constructed to irrigate the nearby Humpty Doo Rice Project (Parks and Wildlife Commission of the Northern Territory 2017). Though the project ultimately failed, the introduction of rice and farming equipment to this area does point to other possible origins of this rice paddy-loving species. However, the species does not appear to be as widespread or as abundant at present for it to have been present for more than 60 years in the Top End of Australia where suitable habitats are so prevalent.

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Book Review: *Dinosaurs – How They Lived and Evolved*

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This morning I awoke to the sounds of dinosaurs in the garden. No. I had not been magically transported back in time during my sleep, nor was someone outside watching the latest instalment of *Jurassic Park* on their personal device. It was birdsong … and it was beautiful!

One of the most interesting facts I discovered when reading the book by Darren Naish and Paul Barrett called *Dinosaurs – How They Lived and Evolved* is that about 160 million years ago a group of dinosaurs called therapods (small, feathered, predatory dinosaurs) evolved into birds. The authors present and analyse a considerable body of evidence that proves this hypothesis, so it's no longer correct to say that *all* dinosaurs became extinct – they still walk, fly and 'sing' among us every day. Some of us even have them as pets!

The authors are world-leading experts on dinosaurs and they provide a highly detailed account of the evolutionary relationship between dinosaurs and explain how the understanding of this important field is being built upon with each new fossil discovery, data analysis technique, theory, climate modelling enhancement and advancement in technology. Following their 2016 (hardback) book of the same title, this new book is officially a 'fully revised and updated' version, but it is really a second edition that includes a new (soft) cover, as well as modifications, updates and corrections.

Tracing the evolutionary path of dinosaurs from 225 million years ago through to the great extinction event 66 million years ago, the book is divided into six chapters: History, Origins and their World; The Dinosaur Family Tree; Anatomy; Biology, Ecology and Behaviour; The Origin of Birds; and the Great Extinction and Beyond.

The authors explain what is understood about the skeletons, organs and muscles of dinosaurs. Of note is that dinosaurs are tetrapods, which is part of the group of vertebrates and have similar skeletal structures to humans. As a reader possessing a basic knowledge of the human skeletal structure and an understanding of the spinal cord and the central nervous system, the chapter on the anatomy of dinosaurs was particularly interesting for me. It also discusses the skin and soft tissue of dinosaurs and applies well-reasoned logic to explain their facial features and functions.

Presenting a significant body of evidence and applying increasingly advanced scientific techniques, the authors examine the things that we take for granted when we are able to study living specimens – their feeding behaviour, movement, reproduction, growth rates and interactions with other species. Visualising a 7 m dinosaur preying on smaller

ones some 2–3 m in length is made possible with a very clear description of the 'grab and bite' theory. Equally interesting is the comparison to the technique used by hawks and falcons, whereby a dinosaur would restrain its prey with large, powerful clawed feet – with the predator eating its prey alive whilst perched on a vantage point.

Having lived and evolved over a considerable time, many species of dinosaur came and went throughout the period covered in the guide. Many of those that appeared earlier were significantly different to those that appeared in later time periods. Dinosaurs grew much bigger, their body shapes changed, diets varied, behaviours changed, some moved on four limbs, others on two and, as explained earlier, some even flew.

The guide discusses the evolution of feathers and other bird-related characteristics and presents the latest research and discoveries of the small therapods of the late Jurassic and Cretaceous periods – and the branch which evolved into modern birds.

Without living specimens to study, the authors hypothesise on their life histories, growth and reproductive behaviour through the extensive interpretation and application of direct fossil evidence and the behaviours of living animals. Given the size and shapes of some of the more well-known dinosaurs such as *Tyrannosaurus rex*, the three-horned *Triceratops* and the plate-backed *Stegosaurus*, it's fascinating to think of the physical interactions required that would enable them to get physically close enough to successfully mate.

In discussing the science underpinning the great extinction of 66 million years ago, the authors discount the many earlier hypotheses as to why this event occurred, including climate change, disease, fungal infection, inability to adapt, and parasites (even ravenous caterpillars!). Presenting evidence from the 1980s onwards, the authors explain the significance of different geological research and analysis which supports the asteroid impact hypothesis and the significant impact this had on dinosaurs. In discussing the 'great extinction' the authors also present evidence to demonstrate that the level of diversity among dinosaurs had been declining prior to the asteroid impact – indicating that habitat change, habitat loss and sustained volcanic activity all contributed to severe disruptions in breeding and migration cycles and this was accompanied by inconsistent plant growth across many regions. The authors note that the reason why *all* dinosaurs did not become extinct is yet to be answered satisfactorily – but they present a number of suggestions why species belonging to several bird groups managed to survive. These include their size and mobility, ability to fly and relocate to new areas, lower dietary requirements, and also that many that survived were largely inhabitants of the southern hemisphere where the effects of these major events may not have impacted as significantly as the northern hemisphere.

This guide to dinosaurs and how they lived, evolved and (largely) became extinct is extremely well written, researched and illustrated. It presents historical and recent evidence demonstrating how fossil discoveries, combined with enhancements in science and technology, have shaped our knowledge and understanding of how dinosaurs dominated the land environment for more than 160 million years. Using photographs,

artistic reconstructions, digital modelling and drawings, the illustrations throughout the book are incredibly striking and allow the reader to clearly visualise how dinosaurs lived – and the specific behaviours, anatomy and biology of each species.

From the perspective of someone without a science background or an in-depth knowledge of contemporary ideas about dinosaurs, understanding this book did require re-reading of text, regular checking of scientific terms and use of the extensive glossary. But in no way did that detract from the quality and enjoyment of what is a very informative, factual, educational and comprehensive guide.

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Book Review:
Australian Birds of Prey in Flight

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With the plethora of bird field guides and the relative affordability of high quality binoculars and cameras these days, identifying birds has never been easier for the lay person. The problem is that birds don't always display co-operatively in plain sight and the only glimpse one may get of a bird is a flash of feathers before one's eyes or a silhouette soaring in the distance. This is especially the case with birds of prey, which are most visible in flight as a speck on the landscape. Thankfully, ornithologists Richard Seaton, Mat Gilfedder and Stephen Debus have collaborated to produce a photographic field guide to assist with the identification of one of the most challenging groups of birds. Their book, just published, is called *Australian Birds of Prey in Flight.*

The idea for this book came during the planning of surveys for Red Goshawks in Queensland. I suspect this was due to some dubious Red Goshawk records in places such as central Australia, where a rufous-morph Brown Falcon or young Black-breasted Buzzard may have been mistakenly identified as this rare raptor that is usually confined to tall coastal tropical and subtropical forests and woodlands. With birdwatching increasing in popularity as a hobby and people eager to contribute data, especially on rarer birds, a need has been identified for birdwatchers to become better informed on the nuances of identifying birds of prey in flight.

The format of the field guide is pleasingly simple. Important to note at the start is a diagram that presents the upper and lower body of a bird of prey, and the terms used for different parts of the body that can be seen in flight. Whilst some of these terms are technical and rarely used (i.e. carpal bar, underwing coverts), they are later referred to in species profiles and aid identification. A four-page species profile is provided for every Australian diurnal bird of prey, as well as the two vagrant species. The first two pages consist of colour images of the species at a variety of distances and angles overlaid on typical habitat in which the species is likely to be found. The third page displays the species at six standardised angles to illustrate the features of flight that aid identification of the species.

The fourth page of the species profile contains the text, consisting of a brief description of the species, other species with which it is readily confused, key identifying features, and a distribution map. The Key Identification Points summary encompass an impressive level of detail, but it might be overwhelming for the novice bird enthusiast. Much of the technical information relating to shape, proportions and plumage will be useful to the birdwatcher who has managed to catch a photograph of a species that he or she is unable

to identify immediately. The most crucial component of this section is the Primary Distinguishing Features paragraph, as this describes the most prominent characteristics that will distinguish the featured species from other birds of prey. Birdwatchers should get familiar with these features when identifying birds of prey in flight.

The final part of the book displays similar looking species alongside each other for quick cross-referencing. This is a useful tool as it can be relatively easy to whittle down the list of potentially species to two or three, and a quick look at the Species Comparison page could provide the definitive identification. More emphasis could have been placed on the habitats of different species, as birds of prey that look similar in flight, such as Grey Goshawk and Grey Falcon, tend to occur in very different habitats and can be differentiated on this basis. However, I understand the authors' reluctance to broach this aspect due to the potential to confound the identification process.

Australian Birds of Prey in Flight is a useful addition to the reference library of any keen naturalist. Whilst the book contains an impressive array of photos it's not designed to sit on a coffee table. It's intended to be taken out into the field and quickly whipped out from a backpack as a bird of prey fades from the birdwatcher's field of vision. The book is astutely designed for simplicity of use, whilst containing an impressive level of technical detail that only a raptor boffin such as Stephen Debus can provide. With the rise of ornithological data gathering via citizen science surveys, a need has arisen to hone the skills of birdwatchers throughout Australia. This field guide will definitely help achieve that task.

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Seaton, R., Gilfedder, M. & Debus, S. (2019 = both paperback edition and ePDF edition) *Australian Birds of Prey in Flight.* CSIRO Publishing, Clayton, Victoria.

Book Review: *A Field Guide to the Freshwater Fishes of the Kimberley*

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I do not know a lot about fish so I was a little surprised to be asked to review *A Field Guide to the Freshwater Fishes of the Kimberley* written by seven fish scientists and a linguist. But perhaps perversely I do not need to have this knowledge, because this is a field guide written for amateurs. It covers that part of Western Australia north from the Fitzroy River and extends into the Northern Territory to the Victoria and Fitzmaurice Rivers. It includes 65 freshwater species and recognises the existence of several undescribed species. In addition there are a further 33 estuarine or marine species included.

Personally, I was disappointed by a number of impressive-sounding introductory topics including evolution, biogeography, climate change and several on understanding indigenous languages. These fall short of expectations for a number of reasons and should have been completed, renamed or omitted. This is a pity but it does not detract from the value of the book.

Families are arranged into one of three groups: Elasmobranchs or cartilaginous fish; Teleosts or bony fish; and (a small section on) Marine-Estuarine Vagrants with much briefer text, but one needs to be an ichthyologist to understand the family sequences here. Each family is given a brief yet informative introduction. Species at least are arranged alphabetically by scientific name within families.

The majority of the book is devoted to species descriptions, most with at least one photograph and a distribution map with text under the headings of General Features, Recognition, Biology and Habitat, Distribution and Abundance, Conservation Notes, Local [indigenous] names and [reference] Sources for each species.

I like this book and I highly recommend it. It fills a geographical niche and includes some Northern Territory fishes also covered by the long out-of-print *Freshwater Fishes of the Northern Territory* by Helen Larson and Keith Martin (1990). Individual species treatments are excellent, as is the glossary and the illustrated guide to families.

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Advice to authors

The *Northern Territory Naturalist* publishes works concerning any aspect of the natural history and ecology of the Northern Territory or adjacent areas of northern Australia. It is a registered, peer-reviewed journal (ISSN 0155-4093) for original research. Contributors include a range of field naturalists and scientists who do not have to be members of the Northern Territory Field Naturalists' Club Inc.

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